SOURCES AND EFFECTS OF IONIZING RADIATION

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ANNEX E

EFFECTS OF IONIZING RADIATION ON NON-HUMAN BIOTA

CONTENTS

INTRODUCTION

A. Background

1. The estimation of human exposure to ionizing radiation from radionuclides of natural and artificial origin is an important and ongoing function of the Committee. The Committee has used simplified generic models of the dispersion and transfer of radionuclides through the environment to estimate the internal and external exposure of humans and the resulting doses. Owing to the complexity and interactions of the underlying processes, special attention has been given to radionuclide transfer via human food chains and the assessment of ingestion doses. The underlying model assumptions and parameters are kept under review and revised as necessary. The last revision was documented by the Committee in annex A, "Dose assessment methodologies" of the UNSCEAR 2000 Report [U3].

2. In the past decades, scientific and regulatory activities related to radiation protection focused on the radiation exposure of humans. The prevailing view has been that, if humans were adequately protected, then "other living things are also likely to be sufficiently protected" [I8] or "other species are not put at risk" [I5]. Over time, the general validity of this view has been questioned on occasion and therefore consideration has been given to the potential effects of exposure to ionizing radiation of non-human biota. This has occurred, in part, as a result of the increased worldwide concern over the sustainability of the environment, including the need to maintain biodiversity and protect habitats and endangered species [U22, U23]; in part, because it has increasingly been recognized that the exposure scenarios and pathways for assessing human exposure may not apply to non-human biota; and, in part, as a result of various efforts to assess the effects of exposure to ionizing radiation on plants and animals [C1, D1, F5, I1, I2, I3, I4, I9, N6, P13, R9, T1, W16].

3. The Committee initially addressed the effects of radiation exposure on plant and animal communities in a scientific annex, "Effects of radiation on the environment", of the UNSCEAR 1996 Report [U4]. Prior to this, the Committee had considered living organisms primarily as part of the environment in which radionuclides of natural or artificial origin may be present and contribute to the internal exposure of humans via the food chain. Like man, however, organisms are themselves exposed internally to radiation from radionuclides that have been taken up from the environment and externally to radiation in their habitat. In general terms, the Committee, in its 1996 report, considered that population-level effects were of primary interest and, of those, that reproductive effects were the most sensitive indicator of harm. Furthermore, it also concluded that it was unlikely that radiation exposures causing only minor effects on the most exposed individual member of a population would have significant effects at the population level; that chronic exposures to low-LET radiation at dose rates of less than 100 mGy/h to the most highly exposed individuals would be unlikely to have significant effects on most terrestrial animal populations; and that maximum dose rates of 400 mGy/h to a small proportion of the individuals in aquatic populations of organisms would not have any detrimental effects at the population level.

4. The International Commission on Radiological Protection (ICRP), the International Atomic Energy Agency (IAEA) and other international organizations have encouraged the exchange of information on the effects of radiation exposure on non-human biota [I19, N6]. The IAEA's action plan on the protection of the environment was discussed at the 2003 Stockholm Conference [I1], which concluded that "While accepting that there remain significant gaps in knowledge and that there needs to be continuing research … there was an adequate knowledge base to proceed and (the Conference) strongly supported the development of a framework for environmental radiation protection". It also found that "the time is ripe for launching a number of international initiatives to consolidate the present approach to controlling radioactive discharges to the environment by taking explicit account of the protection of species other than humans".

5. In 2000, the ICRP, recognizing that environmental protection is a global matter, set up a Task Group to examine the issues. It considered that an approach to environmental protection from ionizing radiation "should relate as closely as possible to the current system for human radiological protection, and that these joint objectives could therefore best be met by the development of a limited number of Reference Animals and Plants" [I9]. Subsequently, the ICRP decided to establish a new Committee (ICRP Committee 5) on the Protection of the Environment. The ICRP further noted that "as radiation effects at the population level—or higher—are mediated via effects on individuals of that population, it seems appropriate to focus on radiation effects on the individual for the purpose of developing a framework of radiological assessment that can be generally applied to environmental issues" [I10].

6. Since the preparation of the UNSCEAR 1996 Report [U4], the approaches to evaluating radiation doses to non-human biota have been reviewed and improvements made [C1, E1, F1, F5, U26]. Information on the levels of radiation

exposure below which biological effects are not expected or, alternatively, above which such effects might be expected, has been developed. This has been obtained, in part, for the projects on the Framework for Assessment of Environmental Impact (FASSET) [F1] and the Environmental Risk from Ionising Contaminants: Assessment and Management (ERICA) [E1], in particular, as part of the development of the FASSET Radiation Effects Database (FRED) [F3]. This information was subsequently integrated with the database on the effects of radiation exposure from the project on Environmental Protection from Ionising Contaminants in the Arctic (EPIC) [B26] resulting in the so-called FREDERICA database [F20].

B. Scope of annex

7. The scientific information given in the FRED [F20] combined with that obtained in the subsequent ERICA programme [G11, J6] and that from more recent studies, especially those undertaken around the site of the Chernobyl accident, provided the basis for the Committee's review of the effects of exposure to ionizing radiation on non-human biota given in this annex. In particular, the Committee used the information from its review to re-evaluate its recommendations on dose rates below which exposure to ionizing radiation is unlikely to result in detrimental effects on populations of non-human biota, given in the UNSCEAR 1996 Report [U4].

8. This annex only provides the Committee's overview of the current data and methods to assess doses to non-human biota and a brief discussion of the nature of effects of radiation exposure on individual organisms and populations. Detailed discussion of these topics is beyond the scope of this annex.

C. Effects of exposure to ionizing radiation

9. Since the preparation of the UNSCEAR 1996 Report [U4], a number of radiobiological phenomena have been described, including genomic instability (genomic damage expressed post irradiation after many cell cycles) and the bystander effect (whereby non-irradiated cells in proximity to irradiated cells exhibit effects similar to those seen in the irradiated cells). These phenomena were discussed in annex C, "Non-targeted and delayed effects of exposure to ionizing radiation", of the UNSCEAR 2006 Report [U1]. While such phenomena are relevant to understanding mechanisms for the development of effects on non-human biota after exposure to ionizing radiation, a discussion of such phenomena is beyond the scope of this annex.

10. The immediate effects of ionizing radiation exposure may be seen at various levels of organization from the subcellular through individual organisms to populations and ecosystems [G16]. Responses of various biological functions to radiation exposure (e.g. reproductive success, metabolic impairment and changes in genetic diversity) can be traced to events at the cellular or subcellular level in specific tissues or organs.

1. Individual level effects

11. Even though mutational events in somatic cells are primarily responsible for cellular transformation and tumour formation, the occurrence of cancer in individual organisms is normally of low relevance to the ecosystem as a whole, except in the case of endangered or protected species [A13]. However, mutational effects in germ cells may lead to reproductive impairment [A14]. Genotoxic stressors, including ionizing radiation, may alter reproductive success by decreasing fertility via clastogenic and mutagenic effects in germ cells resulting in a decrease of the number of gametes. Such stressors may also increase the frequency of developmental abnormalities, e.g. when mutations are induced in germ cells and the progeny of exposed parents develop abnormally.

12. There are a number of weaknesses in the data on which to base estimates of the dose rates below which effects on non-human biota are not considered likely. In addition, there are also issues in extrapolating from the effects observed at cellular and subcellular levels to effects that might be observed in individual organisms, populations and ecosystems. Moreover, it is only under controlled conditions in the laboratory that organisms can be exposed to a single stressor. This presents a further source of uncertainty in extrapolating the results to real ecosystems where multiple stressors exist. Although beyond the scope of this annex, the Committee acknowledges that improved understanding of the mechanisms of radiation damage, of how to extrapolate information from lower to higher trophic levels, and of the possible consequences of multiple stressors is of great interest and worthy of further study.

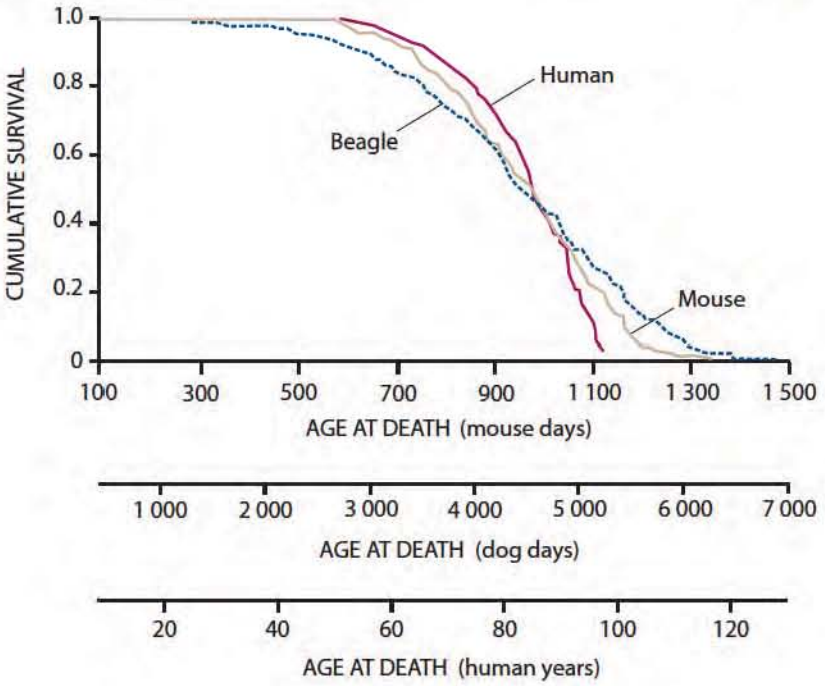
13. The scientific literature provides many examples of adaptive responses to and hormetic effects of exposure to ionizing radiation. Annex B of the UNSCEAR 1994 Report [U5] provided a comprehensive discussion of adaptive responses. In that report, the Committee concluded that there was evidence of an adaptive response in selected cellular processes following exposure to low doses of low-LET radiation but went on to suggest that it was premature to conclude that adaptive cellular responses had beneficial effects that outweighed the harmful effects of exposure. Subsequent to the UNSCEAR 1994 Report [U5], there have been numerous papers and considerable discussion concerning the possibility of hormetic responses to low doses of gamma radiation. For example, Boonstra et al. [B39] reported possible hormetic effects of gamma radiation exposure on populations of meadow voles. These authors suggested that increases in glucocorticoid levels associated with chronic gamma irradiation at a rate of about 1 mGy/d may be an important factor in the increased longevity of exposed meadow voles compared to non-exposed ones. Mitchel et al.

[M9] found that a single dose of 10 mGy to radiation-sensitive mice (Trp53 heterozygous) reduced the risk of both lymphoma and spinal osteosarcoma by greatly delaying the onset of malignancy. Further discussion of adaptive responses and potential hormetic effects of low dose and low dose-rate gamma radiation exposure is beyond the scope of this annex.

14. The various life stages of organisms differ in their sensitivity to exposure to ionizing radiation. It is often assumed that a population will be protected if the most sensitive stage of the life cycle is protected. For a large number of stressors, this assumption seems to be widely true [F9]. However, the most sensitive life stage is often difficult to identify a priori. Consequently, if data on effects only exist for one or two life stages, it may not be possible to know for certain if these data represent information for the most sensitive life stage, even though most of the available information indicates that gametogenesis and embryonic development are among the most radiosensitive stages of the life cycle [I4]. For example, Anderson and Harrison [A15] showed that the synchronous spawning in polychaete worms rendered the organisms susceptible to low-level cumulative impact of ionizing radiation exposure. Because they spawned synchronously and died, oocytes were formed all at once, and damaged gametes could not be replaced.

15. The propagation of effects on individuals to the population as a whole depends greatly on the characteristics of the specific life history. The relative importance of each stage in the life history also varies between species, depending on the specific reproductive characteristics (short generation time versus long generation time, iteroparous versus semelparous, sexual versus asexual reproduction, etc.). Changes in the value of an individual parameter such as age of reproduction (i.e. generation time) often have much stronger consequences for species with fast population growth rates (i.e. with short generation time and high fecundity rate) than for those with slow population growth rates [G3]. On the other hand, the National Council of Radiation Protection and Measurements (NCRP) [N8] noted that when natural causes of deaths are considered collectively on a biologically comparable time scale, natural mortality occurs at a biologically comparable age, as illustrated in figure I.

Figure I. Cumulative survival curves of the mouse, beagle and human for natural causes of death



2. Population and ecosystem level effects

16. Whatever the stressor considered, population-level effects are valuable indicators of ecological hazard (e.g. [F9]). However, because of experimental constraints, most available data describe the effects on the individual traits of irradiated organisms. Many studies have documented the effects of radiation exposure at the cellular, tissue and individual levels. The consequences have been found to be increases in morbidity and mortality, decreases in fertility and fecundity, and increases in mutation rate [W10]. These types of effect, observed at the individual level, may have consequences for a population of a species.

17. Matson et al. [M12] and Baker et al. [B29] investigated the possible genetic and population effects resulting from the chronic radiation exposure of bank voles, *Clethrionomys glareolus*, inhabiting contaminated sites near Chernobyl.

Both groups reported that genetic diversity was elevated in the contaminated sites when compared to relatively uncontaminated sites but were unable to attribute any significant detrimental effects among the bank vole populations to radiation exposure.

18. Ionizing radiation does not appear to have any direct effects at the population or higher ecological levels (i.e. community or structure and function of ecosystems). At present, it appears that all such effects are mediated by effects at the individual or lower levels. In addition, indirect effects through food-web mediated processes may occur [G16]. One approach to extrapolating from the effects on individuals to effects at the population level is to integrate the effects on survival and reproduction in terms of population growth rate. Population growth rate is one of the most important characteristics of a population and is defined as the population increase per unit time divided by the number of individuals in the population. Population models are used to extrapolate from the toxic effects on individuals, expressed as modifications to values of life-cycle parameters, to effects at the population level. This method has been used, for example, by Woodhead [W10] in a theoretical way and was implemented through experiments within the ERICA project for the chronic exposure of two invertebrates exhibiting contrasting life cycles: the earthworm and the daphnid [A26, G3].

19. An ecosystem has complex interactions between biotic and abiotic components and among biotic components. The latter are called interspecific interactions and include competition, predation and association. These interactions contribute to the flow or cycle of energy, materials and information in the ecosystem, and thus provide the ecosystem with its fundamental property of self-organization. It is possible that if one species is directly damaged by a toxic agent, another species more resistant to that agent is also indirectly affected by the depletion of interactions with the directly damaged species. As a result, the entire ecosystem can be affected in extreme cases. These indirect effects have been observed in ecosystems exposed to ultraviolet radiation [B37] and some chemicals [C23, H24, M24, T24, W20]. Similarly, some indirect effects through inter-species interactions have been observed in irradiated ecosystems, as reviewed in the UNSCEAR 1996 Report [U4]. Given this backdrop, the importance of indirect effects has been considered in reviews of the effects of exposure to ionizing radiation on ecosystems [B38, C21, I2, I3, I4, N1, U4]. Since these indirect effects cannot necessarily be deduced from effects on individuals and populations, ecosystem-level effects are evaluated using mathematical modelling, model ecosystem experiments and field irradiation experiments.

3. Multiple stressors

20. In general terms, the modifying effects of multiple stressors can be considered in one of two broad categories, namely *(a)* the modification by the other stressors of the organism's uptake of radioactive material and the distribution of radioactive material within the organism, and *(b)* the influence of the other stressors on the radiosensitivity of the species [A18, B28, F5, G18, L8, P9, R19, S17, S18].

21. Metabolic manifestations of exposure to ionizing radiation include impairment in enzyme function, altered protein turnover, impairment in general metabolism and inhibition of growth. Sugg et al. [S17] showed that the body condition of largemouth bass exposed to mercury and ^{137}Cs in different lakes near the Savannah River site could be related to DNA damage. Changes in lipid metabolism in fish liver and a stimulation of the ventilation rate of a lamellibranch species have also been shown to occur at low doses in this mixed exposure scenario [P22, P23].

22. Experiments involving multiple exposures to metals (cadmium and zinc), organic pollutants, such as polychlorinated biphenyl (PCB), polycyclic aromatic hydrocarbon (PAH), endocrine disruptors, and radionuclides (radioactive isotopes of cobalt, caesium, and silver) have been conducted both under controlled conditions and in the field [G17]. Experiments using a freshwater bivalve (*Dreissena polymorpha*) and a carnivorous fish (*Oncorhynchus mykiss*) exposed under chronic conditions to water containing concentrations of 1–4 μg/L of cadmium and/or 170–250 μg/L of zinc showed a 60% decrease in the bioaccumulation of the isotopes of silver and caesium in the bivalve and a 30% decrease in the fish. However, no effect was observed for other radionuclide/organism pairs (such as cobalt for the fish). On the other hand, prior exposure to organic micropollutants enhanced both the uptake and retention of ^{57}Co and ^{134}Cs in the fish. Several possible explanations, linked to a modification of the health status of the animal by the presence of stable pollutants, were advanced by the authors and supported by biomarker measurements: an increase in respiratory activity by alteration of the global metabolism; a decrease in the Na^+/K^+-ATPase in gills and therefore modification of the ionic flux; or an alteration of the epithelium permeability [A16, A17, F15].

23. Genotoxic/cytotoxic damages are not specific to ionizing radiation and may also be initiated by other toxins [S18]. Indeed, most biochemical techniques for detecting DNA damage at the molecular or cellular level lack specificity for radiation-induced DNA damage [T9]. However, Tsytsugina [T8] and Tsytsugina and Polikarpov [T6] analysed the distribution of chromosome aberrations in cells and the frequency of the different types of aberrations in order to discriminate between the contributions of radiation and chemical factors to the total damage to natural populations in aquatic organisms. These studies showed that the chromosome damage observed in aquatic worm populations exposed to dose rates of 10 μGy/h or more in lakes located in the vicinity of the site of the Chernobyl accident was mainly caused by radioactive contamination. Hinton and Bréchignac [H20], however, cautioned that, while there is a great potential value in using biomarkers for assessing risks to non-human biota, there remain many challenges in linking changes in biomarkers at

the molecular or cellular levels to effects on individual organisms and populations of organisms.

24. The antioxidant status modified by exposure to various stressors may influence the radiosensitivity of organisms. The cellular damage due to radiation exposure is mainly associated with oxidation. This oxidative stress may also be caused by other stressors, such as chemical pollutants, and cellular defence mechanisms against reactive oxidative species (ROS) that may be solicited are not stressor specific [S27]. Therefore, the interaction of heavy metals and radionuclides, and the resulting modification of radiosensitivity, may depend on the capability of the antioxidant defence systems of the organism [C13, C14, C15, S27, V1].

25. The potential effects of exposure to uranium in the environment may arise from the chemical toxicity of the metal and its radiotoxicity (arising from the uranium alpha particles) and thus, such situations can be regarded as being due to a mixture of stressors coming from a single element [B30, C19, P24]. Thus, while an evaluation of the chemical toxicity of uranium to non-human biota is beyond the scope of this annex, it is important to recognize that the chemical toxicity and the radiological effects of uranium occur concurrently, and that both may need to be considered in a practical assessment of risks to non-human biota.

4. Commentary

26. Most of the data on the effects of exposure to ionizing radiation on non-human biota are from observations made on individual organisms. Radiation effects on populations occur as a result of the exposure of individual organisms. The propagation of effects from individual organisms to populations is complex and depends on a number of factors. However, as suggested in the UNSCEAR 1996 Report [U4], the most important effects appear to be those on reproduction and reproductive success. Many questions remain with respect to the following: the mechanisms whereby radiation exposure can cause harm; inter-species extrapolation; propagation of harm from nuclear DNA to the population; and the effects of multiple stressors. Moreover the possibility of hormetic effects at low doses and dose rates of gamma radiation, the relation between changes in biomarkers at the molecular and cellular level and the effects on individual organisms or populations of organisms, and the effects of multiple stressors continue to be of considerable interest.

D. Observations from case studies

27. Ecological risk assessments (ERAs) have been conducted for a wide variety of situations where non-human biota are exposed to enhanced levels of radiation or radioactive material. ERA studies are available for a wide variety of nuclear fuel cycle activities from uranium mining to waste management, as well as for sites with enhanced levels of naturally occurring radioactive materials, and for sites contaminated as a result of accidents. Table 1 outlines the key elements of an ERA framework for assessing the effects of exposure to ionizing radiation on non-human biota. Various approaches for performing ERAs have been outlined including those of the IAEA [I2, I3, I4], NCRP [N1], the United States Department of Energy (DOE) [U26], Jones et al. [J1], Environment Canada and Health Canada [E2], FASSET [F1, L4] and ERICA [B17]. All of the approaches necessarily involve simplifications of the knowledge about the actual environment. A common approach to the assessment of the effects of radiation exposure on non-human biota involves the use of a screening index (*SI*), where *SI* is simply a dimensionless ratio of the estimated dose rate (to an individual organism) to the reference radiation dose rate, viz.:

$$SI = \frac{\text{estimated dose rate}}{\text{reference dose rate}} \quad (1)$$

Table 1. Key elements of a framework for the assessment of the effects of radiation exposure on non-human biota

Element	*Considerations*
Exposure of biota	• Spatial and temporal patterns of radionuclide concentrations in environmental material • Uptake by organism • Non-uniform distribution within organism
Reference biota	• Not possible to evaluate all biota • Need to select reference biota or indicator species appropriate for area of interest and desirable basis for selection • Possible need to consider individual biota per se when species are endangered
Dosimetry model for (reference) biota	• Absorbed dose (to whole body or to tissue/organ) • Geometry corrections • Relative biological effectiveness (RBE): the effects of different qualities of radiation on biota
Endpoints in radiological assessment	• Selection of appropriate population-level (deterministic) "umbrella" effects such as mortality or reproductive capacity and corresponding reference doses
Effects on biota	• Connection between radiation effects on "umbrella" endpoint in individual, and consequent "possible" effects on population • Role of background radiation levels • Natural population variability

28. The reference dose rate refers to the chronic dose rate (commonly expressed in milligray per day) below which potential effects on populations of organisms are not expected. The ratio, *SI*, assumes that the estimated dose rate and the reference dose rate relate to the same endpoint (e.g. mortality, reproductive capacity). The estimation of dose rate to an individual organism is discussed in section I of this annex. As there are many complex factors involved, caution is needed in extrapolating from the effects of radiation exposure on an individual organism to those on a population of organisms [B17].

29. The reference radiation dose rates for particular endpoints developed by the Committee in the UNSCEAR 1996 Report [U4] have been the most commonly used for the denominator of the *SI* calculation. However, other guidance has also been developed [C1, E1, E2, F5, I4, N1] and, more recently, the concept of species sensitivity distributions (SSDs) has been introduced [B17, G3]. These developments may necessitate a re-evaluation of the reference dose rates obtained in the ERA case studies.

30. Because of the sparsity of peer-reviewed literature, all of the various sources of information on reference dose rates (e.g. various reports and supporting environmental assessments in Canada, technical reports of government agencies in various countries and conference proceedings) have been considered in this annex.

31. Of the numerous reports [A24, A25, B17, C1, C2, C20, C22, E2, E3, E5, E22, E23, F2, G2, G3, G27, J2, S10, S11, S32, S33, U26, W19], only a few provide studies of the radiation exposure of non-human biota arising from radioactive waste management activities or accidents involving dose rates close to or exceeding the reference dose rates [A25, E8, E22]. For example, one study [S39] which involved investigation of the risks to biota from exposure to ionizing radiation from nuclear fuel cycle activities in Canada concluded that the largest risk is associated with past uranium mining activities; that discharges of radioactive material from power reactors under normal operating conditions are not expected to cause environmental harm; that organisms within one of the waste management areas examined may be harmed by exposure to ionizing radiation; and that current radioactive discharges from uranium refineries and conversion plants are not expected to cause environmental harm. Similar results can be derived from a consideration of the case studies reported in ERICA [B17] of a wide variety of nuclear fuel cycle and other activities.

32. One study in which the estimated dose rates to biota exceeded the reference dose rates, at least over a limited area, was of the radioactive waste management site at the Chalk River Laboratories (CRL) located on the shore of the Ottawa River, 160 km north-west of Ottawa, Ontario, Canada [E23]. The CRL site was established in the mid-1940s and has a history of various nuclear operations and facilities, primarily related to research. An ERA was conducted to assess the doses to biota arising from elevated levels of tritium, ^{14}C, ^{41}Ar, ^{90}Sr, ^{131}I, ^{137}Cs and ^{239}Pu and from radionuclides that are naturally present in the environment, for example, the uranium series radionuclides, using standard methods for evaluating the uptake of these radionuclides by biota from the affected aquatic and terrestrial environments [B12]. A reference dose rate of 1 mGy/d was used for all organisms [B36]. Dose rates to some aquatic organisms such as frogs, small fish, snails and aquatic plants within the on-site waste management areas were estimated to be above the reference dose rate of 1 mGy/d; however, outside of the actual waste management areas, dose rates were estimated to be below the reference dose rate. The main contributor to the estimated dose rates to invertebrates and terrestrial plants was ^{90}Sr in surface soil, while that to the woodchuck (estimated at 51 mGy/d) was inhalation in the burrow of ^{222}Rn decay products from background levels of ^{226}Ra in the soil. A few individual invertebrates and terrestrial plants actually within the confines of small on-site waste management facilities were also estimated to have been subjected to dose rates above 1 mGy/d. Based on the limited spatial extent of the estimated dose rates that exceeded the reference dose rate and environmental observations, the authors considered that significant effects at the population level were unlikely.

33. Much of the new information on the effects of exposure to ionizing radiation on organisms has arisen from studies in the area surrounding the site of the Chernobyl accident, where dose rates to organisms were above the reference dose rate suggested in the UNSCEAR 1996 Report [U4]. A summary of the results of these studies up to 1996 is provided in this annex. Section III of this annex provides a comprehensive review of the more recent data from studies of non-human biota in the area surrounding the site of the Chernobyl accident.

E. Structure of this annex

34. The prime purpose of this annex is to build on the information reported in the UNSCEAR 1996 Report [U4]; to compile data that has since become available on the effects of exposure to ionizing radiation on non-human biota; and to determine if the reference dose rates need to be updated. However, it is necessary first to provide some general information on the relationships between the levels of radiation in the environment in which the biota live and the consequent dose (or dose rate) to biota as a whole or selected tissues and organs. Table 1 provides a summary of five key elements that form the basis for assessing the effects of exposure to ionizing radiation on non-human biota.

35. The relationships between the levels of radiation exposure and the activity concentration of radioactive material in the environment and the dose to an organism living in that environment is the subject of section I.

36. Section II provides a summary of the information considered in the UNSCEAR 1996 Report [U4] and the key observations from that report.

37. Section III provides an overview of the findings of the studies of non-human biota in the area surrounding the site of the Chernobyl accident. It includes the work of the Chernobyl Forum [E8].

38. Section IV provides a summary of the effects of exposure to ionizing radiation on non-human biota derived from the material given in earlier sections and reviews carried out by other scientific organizations and groups, namely, the IAEA [I4], Bird et al. [B1], the DOE [J1, U26], Environment Canada and Health Canada [E2], Canada's former Advisory Committee on Radiological Protection (ACRP) [A1], the UK Environment Agency [C1], the FASSET group [F1, F5, L1, L4], and the ERICA group [E1, G11, G15]. The published literature was also reviewed.

39. Section V provides an overall summary of the data reviewed and, based on these data, the Committee's evaluation of the dose rates below which effects on non-human biota are not considered likely. A few important areas for potential future study are also noted.

I. ESTIMATING DOSES TO NON-HUMAN BIOTA

40. Data on the effects of radiation exposure on non-human biota have been obtained from experimental studies carried out in the laboratory and in the field. Additional data have been obtained from the results of studies on environments with elevated levels of radiation or of radioactive material resulting from normal operations of nuclear facilities, waste management activities, or accidents. The interpretation of the results of these studies requires an understanding of the relationship between the levels of radiation and the activity concentrations of radionuclides in the various environmental media in which the organism resides, the consequent dose rate to an organism (or a tissue or organ of the organism) that lives in the environment, and the biological effect of interest. For example, radionuclides in the ambient environment may lead to external irradiation and internal irradiation as a result of radionuclides being taken into the organism via inhalation, ingestion, or uptake through its skin or membrane. Empirically determined concentration factors and transfer factors are commonly used to estimate contaminant concentrations in the organism (e.g. expressed for wet or dry weight in units of Bq/kg) from concentrations in the ambient environment (e.g. expressed in units of Bq/kg for sediment or soil, or Bq/L for water). Dosimetric models can then be used to derive, for selected organisms, dose conversion coefficients (DCCs) that relate ambient concentrations to internal or external exposure, as appropriate, and hence to dose.

A. Assessing exposures of biota

1. Choice of reference organisms

41. In view of the enormous variety of living organisms, it would be impossible to consider all species of flora and fauna as part of an environmental impact assessment even for a limited area. Instead, a concept has been developed involving the selection of reference organisms that are representative of large components of common ecosystems and for which models are adopted for the purpose of deriving doses and dose rates to organisms, tissues, or organs from radionuclides in the environment. The results of such dose assessments for these predefined reference organisms will allow a basic assessment to be made concerning the possible biological effects. This approach provides a strategy that allows the modelling effort to be reduced to a manageable level. It further provides information on the exposures of different organisms under varying exposure conditions, which allows the estimation of the impacts on those components of the environment for which data may be sparse or absent.

42. The reference organism approach of the ICRP had its genesis in some earlier publications [P6, P13]. In the framework of the FASSET project [F20, L4], reference organisms were defined as "a series of entities that provide a basis for the estimation of radiation dose rate". The idea was that these organisms would provide a basis for assessing the doses to organisms and consequential effects in general due to radionuclides in the environment. The main criterion for the selection of reference organisms within the FASSET project was that the habitats and feeding habits should be such that the external and internal exposures are maximized.

43. The ICRP is assembling databases that relate to a limited number of "reference animals and plants". These are defined as "hypothetical entities with the assumed basic characteristics of a specific type of animal or plant, as described to the generality of the taxonomic level of family, with defined anatomical, physiological, and life-history properties that can be used for the purposes of relating exposure to dose, and dose to effects, for that type of living organism" [I12].

44. Both the FASSET and the ICRP approaches were intended to simplify the process of estimation and evaluation of exposures to ionizing radiation of non-human biota. Whereas reference organisms in FASSET were specifically selected for different ecosystems (e.g. agricultural, seminatural, freshwater, and marine), ICRP [I10] described the reference animals and plants in groups (family or taxonomic level). The reference organisms selected cover a range of ecosystems and taxonomic families (table 2). The generic (reference) organisms that are explicitly considered in this annex are summarized in table 2. Organisms similar to those adopted by the ICRP were selected for consistency. The features of the selected organisms are described in reference [I10].

Table 2. Comparison of reference organisms defined by different international bodies

Defined by	*Reference organisms*
FASSET Terrestrial ecosystems [L1]	Soil microorganisms Soil invertebrates Plants and fungi Bryophytes Grasses, herbs and crops Shrubs Above ground invertebrate Burrowing mammal Herbivorous mammals Carnivorous mammals Reptile Vertebrate eggs Amphibians Birds Trees
FASSET Aquatic ecosystems [L1]	Benthic bacteria Benthic invertebrates Molluscs Crustaceans Vascular plants Amphibians Fish Fish eggs Wading birds Sea mammals Phytoplankton Zooplankton Macroalgae
ICRP Proposal on Reference Animals and Plants [I10]	Deer Rat Duck Frog Trout Flatfish Bee Crab Earthworm Pine tree Wild grass Brown seaweed
This annex	Earthworm/soil invertebrate Rat/burrowing mammal Bee/above ground invertebrate Wild grass/grasses, herbs and crops Pine tree/tree Deer/herbivorous mammal Duck/bird Frog/amphibian Brown seaweed/macroalgae Trout/pelagic fish Flatfish/benthic fish Crab/crustaceans

2. Radioecological models

45. Three classes of radioecological model can be distinguished and are presented here in terms of increasing complexity—equilibrium models, dynamic models and research models.

46. Equilibrium models are primarily intended for the assessment of exposures due to routine discharges of radioactive material into air or water. They are based on two fundamental assumptions: *(a)* the emission rates of the radionuclides are constant in time; and *(b)* the duration of the discharges is long compared to the time needed for radionuclide transfer

along the environmental pathways considered. With these assumptions, the radionuclide concentrations reach equilibrium within each of the compartments into which the environment is subdivided for modelling purposes, and the transfers between compartments are easily characterized by time-invariant ratios of concentrations between the acceptor and donor compartments.

47. Since equilibrium radionuclide concentrations in the environment are typically attained after considerably long operational times of a nuclear facility, the equilibrium models are likely to give conservative exposure estimates. This type of radioecological model has been used to determine compliance of routine discharges from nuclear facilities with authorized limits [H4, I11, N3, U3].

48. Ciffroy et al. [C22] tested the influence of the time-dependence assumption frequently used in radioecological models in a case study conducted on the Loire River in France. For routine discharges of radionuclides from nuclear power plants, their main conclusions were that: *(a)* attention must be paid to the temporal variations in the discharges, and gaps between actual instantaneous discharges and maximum discharges on a yearly time scale must be analysed; *(b)* the equilibrium assumption at the water-suspended matter interface must be justified and eventually corrected when equilibrium conditions are not expected; and *(c)* for organisms showing slow uptake/elimination rates, a kinetic approach to the bioaccumulation process can avoid some overestimation of radionuclide concentrations. The assumption of equilibrium led to overestimations of one to two orders of magnitude in predicting ^{60}Co concentrations in invertebrates.

49. A number of inherent advantages have contributed to the proliferation of equilibrium models. The model structure can be kept simple, but there is flexibility to allow more detailed structure, if necessary. Under equilibrium conditions, dispersion of trace amounts of radionuclides in the atmosphere or rivers is adequately represented by analytical solutions of more general physical models; transfer via food chains is represented by simple multiplicative chains of concentration ratios.

50. A major conceptual limitation of radioecological models is that many of the parameters involved (e.g. concentration ratios) have to be established empirically. Experience gained during recent decades has amply demonstrated that numerical values of many of these parameters may vary by several orders of magnitude; this has been well documented, for example, for plant–soil relationships of radiocaesium and radiostrontium concentrations [F7, F8, N4]. While for the purposes of screening or environmental protection as may be established by the ICRP or required by a national regulator, representative parameter values can be selected that ensure that the model assessments are conservative, obvious difficulties exist if a realistic assessment of exposures in specific ecosystems is needed.

51. Dynamic radioecological models [M4, S13, W3] are applied if the time dependence of exposures that result from varying or instantaneous releases has to be taken into account. Examples of their use include the assessment of the time-dependent radionuclide concentrations in the environment, such as those resulting from accidental radionuclide releases varying over time, and the simulation of seasonal effects, which are of major importance in terrestrial environments during the first year following deposition of radionuclides after an accidental release [M7].

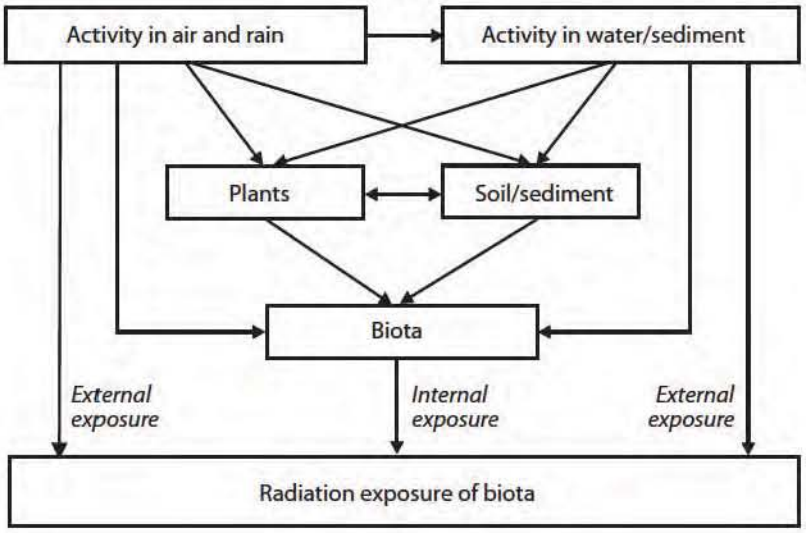
52. Research models are characterized by a high degree of complexity and longer computation times, and presently are limited to simulating a few of the important processes in analyses of environmental pathways for radionuclides [C7, P9]. Currently, therefore, they do not offer an alternative to equilibrium and dynamic radioecological models for environmental assessments, although they do constitute an important tool for improving understanding of the sources of variability observed empirically.

53. The scope of this annex is limited to providing a broad overview of the approach to estimating radiation exposure and subsequent doses to non-human biota. The reader interested in these topics is referred to the extensive literature. Exposure assessments are generally based on equilibrium models. However, for case studies at specific locations contaminated by accidental releases of radionuclides, information on the levels of exposure of local biota taken from the literature is sometimes based on simulations using dynamic radioecological models.

3. Transfer of radionuclides in the environment and resulting exposures

54. The major pathways of radiation exposure of biota in the environment are summarized in figure II. In this schematic representation, the physical components of the terrestrial environment are air, soil and sediment; the biological components include plants, invertebrates, and vertebrates (mammals, birds, reptiles, and land-based amphibians). The physical components of the freshwater aquatic environment include streams, rivers, lakes and sediments; the biological components are phytoplankton, zooplankton, macroinvertebrates, sessile aquatic plants and vertebrates (fish, water-based amphibians and some aquatic mammals). In a marine environment, the physical components include tidal zones, coastal waters and marine sediments; and the biological components include phytoplankton, zooplankton, macroinvertebrates, sessile aquatic plants, and vertebrates (fish and marine mammals), molluscs, crustaceans and marine birds. The terrestrial and aquatic environments are not totally separate. Some birds and terrestrial mammals eat fish and shellfish; moose and waterfowl feed on aquatic plants; and terrestrial animals ingest drinking water from the aquatic environment.

Figure II. Major environmental transfer routes for evaluating radiation exposure of biota



55. The total radiation dose received by an organism (or some organ or tissue of the organism) is the sum of the contributions from both external and internal exposure. External exposure results from complex non-linear interactions of various factors, such as the levels of the radionuclides in the habitat, the geometrical relationships between the radiation source and the target, the shielding properties of the materials in the environment, the size of the organism and the radionuclide-specific decay properties (characterized by the type and energy of the radiations emitted and their emission probabilities).

56. Internal exposure is determined by the activity concentrations of the radionuclides in the organism, the size of the organism, the radionuclide distributions within the organism and the specific decay properties of the radionuclides. In addition, the relative biological effectivenesses (RBE) of alpha, beta and gamma radiation need to be taken into account in assessing the consequences of the exposure.

B. Transfer of radionuclides in the terrestrial environment

57. Radioactive material released into the atmosphere is dispersed and transported by the wind. Exposures of biota are calculated from the activity concentrations of radionuclides in the environmental media, such as air, soils and vegetation, and in the organisms under consideration. The principal processes involved in the transport of radionuclides in the terrestrial environment include dry deposition, wet deposition, interception by vegetation, loss of radionuclides from plants due to weathering, resuspension, the systemic transport of radionuclides within plants, uptake from soil, run-off to water bodies and the transfer to animals. This section discusses the factors that affect the behaviour of radionuclides in a terrestrial environment and the uptake of radionuclides from the environment to plants and animals.

1. Dry deposition

58. Dry deposition per unit time is proportional to the near-surface concentration of the material in air. Usually, the dry deposition of a radionuclide from the atmosphere to soil and vegetation is expressed in terms of the deposition velocity, v_g (m/s), which is defined as the ratio of the activity deposition rate per unit area and the local activity concentration in air of the radionuclide at a reference height. This empirical quantity depends on a variety of factors such as the size of any associated particles, the characteristics of the surface–air interface, the meteorological conditions and the chemical form of the radionuclide.

59. Typical estimates of deposition velocities for grass and forests are summarized in table 3. These values are used for the calculation of the exposures of biota resulting from the atmospheric release of radionuclides.

Table 3. Typical estimates of deposition velocities for grass and forest [P14, R11]

Chemical/physical form	*Deposition velocity (m/s)*			
	Grass	*Forest*[a]		
		Crown	*Trunk*	*Soil*
Particles, 0.1–1 μm	0.001	0.005	0.000 5	0.000 8–0.003
Elemental iodine	0.01	0.05	0.005	0.006–0.02
Methyl iodide	0.000 1	0.000 5	0.000 05	0.000 08–0.000 3

[a] Coniferous trees and deciduous trees with fully developed foliage.

2. Interception of radionuclides deposited from the air

60. Interception defines the fraction of radioactivity deposited by wet and dry deposition processes that is initially retained by the plant. There are several possible ways to quantify the interception of deposited radionuclides. The simplest is the interception fraction, f, which is defined as the ratio of the activity initially retained by the standing vegetation, A_i, immediately subsequent to the deposition event to the total activity deposited. A full description of the interception process is beyond the scope of this annex and the reader interested in this topic is referred to the extensive literature (e.g. see reference [H26]).

61. Radioactive material in air can be washed out by rain and snow. A fraction of the radionuclides deposited with precipitation is retained by the vegetation, and the rest falls through the canopy to the ground. Although the radioactive material retained eventually transfers to soil through weathering and is retained only temporarily by vegetation, the fraction initially intercepted is important owing to the fact that the concentration of radioactive material will be at its highest at this time. Interception of wet deposits is the result of a complex interaction of the amount of rainfall, the chemical and physical form of the deposit and the actual stage of development of the plant [M4] and thus, interception fractions for a single event may vary from 0 to 1.

62. To account for its dependence on biomass in some models, the interception of wet deposited activity is modelled as a function of the biomass density, according to the approach of Chamberlain [C8]. The chemical form is a key factor; since the plant surface is negatively charged, the absorption of anions is less effective than that of cations [H6, H7, K4, M4, P11]. Differences between plants seem to be of minor importance compared to those between radionuclides, e.g. the interception of polyvalent cations is higher than that for anions by as much as a factor of 8 [H5]. However, in general, for the estimation of interception following the routine discharge of radioactive material, very simple approaches are used in the models [P10]. Anspaugh [A22] suggested a default value for the interception fraction of the order of 0.3 for all elements, plants and precipitation events for routine discharges of radionuclides.

3. Weathering

63. Following deposition on vegetation, radionuclides are removed by wind and rain. In addition, the increase of biomass during growth leads to a reduction in the activity concentration. Since growth is subject to seasonal variations, the post-deposition reduction of the activity concentration of radionuclides in plants depends on the season. These processes of reduction in the activity concentration of radionuclides in plants occur simultaneously after deposition. As it is difficult to quantify the exact contribution of each process, the net reduction in the activity concentration with time is usually called "weathering" and expressed by the empirical weathering half-time, T_w.

64. The chemical form of the contaminant seems to be of minor importance in weathering. After the Chernobyl accident, the median weathering half-times observed for iodine and caesium on grass were approximately 8 and 10 days, respectively [K5]. Shorter half-times were observed primarily in regions with fast growing vegetation, while longer half-times were found in Scandinavia, where the growth rates were lower because of the later spring in the area [K5]. In general, longer weathering half-times are observed for slowly growing or dormant vegetation [M8].

65. In forests, weathering is more complex because of the canopy structure, which comprises several vegetation layers, such as crown, trunk and understorey vegetation. Radionuclides lost from the crown may be retained by the understorey vegetation, thus reducing the overall loss rate of radionuclides from vegetation to soil.

4. Distribution of radionuclides within plants

66. The currently available dosimetric models for the assessment of the exposure of biota do not take into account heterogeneous radionuclide distributions within plants. Hence, any information on these distributions cannot currently be used in the assessment.

5. Uptake of radionuclides from soil

67. Soil is the main reservoir for long-lived radionuclides deposited on terrestrial ecosystems. The behaviour of radionuclides in

soils controls their migration in soil, the possible transport to groundwater, and the long-term radionuclide concentration in vegetation and thus the exposure of soil organisms. As for all minerals, the uptake of radionuclides by plants mainly takes place via dissolution from soil. The concentration of radionuclides in soil solutions is the result of complex physical–chemical interactions with the soil matrix, with ion exchange being the dominant mechanism. Ion exchange by its very nature is a competitive mechanism. The concentrations and composition of the major competing elements present in soil thus are of primary importance in determining the distribution of radionuclides between soil, soil solution and plant roots (which are able to influence the microspace in their vicinity in order to provide and maintain conditions that favour the uptake of nutrients) [E6].

68. The physical chemistry of sorption and desorption of radionuclides in the soil–soil solution system and their possible uptake by plants are the result of complex interactions between soil type, pH, redox potential, sorption capacity, clay content, content of organic matter and soil management practice. Although these factors are qualitatively known, they are difficult either to quantify or to integrate into a universal model applicable to a wide range of soil conditions. Consequently, the approaches used include classifying the transfer according to soil types (e.g. peat, sand, loam and clay) and other physical and chemical parameters. In addition, various biological factors should be considered, especially whether or not the radionuclide is an essential element.

69. For the quantification of the root uptake of radionuclides, empirically derived (aggregated and greatly simplified) parameters—soil–plant transfer factors or concentration ratios—are usually applied despite their inherent limitations [E6]. In this case, these parameters are the ratios of the activity concentrations in the plant to those in the soil within the uppermost layer of a standardized thickness. Transfer factors were originally defined for agricultural ecosystems within which radionuclides are distributed homogeneously within the rooting depth of agricultural plants because of ploughing.

70. The aggregated transfer factor is defined as the activity concentration of a radionuclide in a material (Bq/kg) divided by the total deposition—activity per unit area (Bq/m^2)—at equilibrium. The concept of aggregated transfer factors was developed as a simplification of detailed physical and chemical processes to a single value, inter alia, to avoid difficulties with determining radionuclide concentrations in soils with a multi-layered structure, such as in forests.

71. Alternatively, concentration ratios that relate to the activity concentrations of radionuclides in specific soil horizons exploited by the mycelium or the root system were proposed in the late 1980s and proved to be useful, especially in connection with the prediction of the transfer of ^{137}Cs to fungi [G4, R8, Y1, Y4, Y5].

72. Illustrative ranges of soil–plant transfer factors for a number of elements are summarized in table 4 [T11]. This table shows that the uptake of caesium from soil usually does not result in a simple proportional accumulation in plants. Radiocaesium is effectively sorbed by micaceous clay minerals that are present in almost all soils in varying amounts. A detailed compilation of soil–plant transfer factors including data for specific plant groups, plant organs and soil types can be found elsewhere [I14].

Table 4. Typical ranges of soil–plant transfer factors [T11]

Element	*Concentration ratio Bq/kg plant (d.m.) per Bq/kg soil (d.m.)*	*Aggregated transfer factor*[a] *Bq/kg plant (d.m.) per Bq/m^2 soil*
Sr	0.01–1	4×10^{-5}–4×10^{-3}
Cs	0.001–0.1	4×10^{-5}–4×10^{-4}
Cs[b]	0.1–10	4×10^{-4}–4×10^{-2}
I	0.001–1	4×10^{-5}–4×10^{-3}
Tc	0.1–10	4×10^{-4}–4×10^{-2}
Pb	0.001–0.01	4×10^{-5}–4×10^{-5}
Ra	0.001–0.1	4×10^{-5}–4×10^{-4}
U	0.001–0.1	4×10^{-5}–4×10^{-4}
Np	0.001–0.1	4×10^{-5}–4×10^{-4}
Pu	10^{-5}–10^{-3}	4×10^{-8}–4×10^{-6}
Am	10^{-5}–10^{-3}	4×10^{-8}–4×10^{-6}
Cm	10^{-5}–10^{-3}	4×10^{-8}–4×10^{-6}

[a] Calculated from the concentration ratio assuming a mass density for dry matter (d.m.) in the soil rooting zone of 280 kg/m^2 taking account of the mass of the soil within the rooting zone.

[b] Observed range in natural and semi-natural ecosystems on acid sandy soils poor in potassium.

73. Caesium uptake is particularly high from organic soils with a low pH and pronounced potassium deficits [F11]. Such soils are frequently found in the Russian Federation, Belarus and Ukraine, as well as in Scandinavia, the upland areas of the UK and the alpine areas of Europe. For organic matter, the cation exchange capacity decreases with increasing acidity owing to the saturation of carboxyl groups with hydrogen ions. Furthermore, the availability of caesium for uptake is enhanced in soils that are poor in potassium. Additionally, the clay content of organic soils is low and this prevents strong sorption and leads to persistently high caesium levels in plants [A7, F12, F13, K6]. Another important aspect is that the bioavailability of radionuclides and their uptake after deposition may change with time. This was observed in areas close to the site of the Chernobyl accident and was caused by the degradation of fuel particles, the fixation of caesium within the soil and changes in the sorption strength of the soil for caesium [N5, S14, S15].

74. In recent years, a number of experiments have been performed to determine soil–plant transfer factors for tropical and subtropical environments [C9, F11, R6, T12, T13, U24, U25, W12, W13]. The anaerobic soil conditions in flooded paddy fields change the solubility of some elements, such as I and Tc, and thus possibly their soil–plant transfer factors [M25, T26, Y3]. In general, however, the results do not indicate any systematic impact of climatic conditions on the transfer of radionuclides from soil to plants, although the numbers of data are still small. Further data on the tropical and subtropical environments are therefore needed [M25].

75. In forest ecosystems, the transfer of radionuclides from soil to plants and fungal fruit bodies depends on the depth profile of the radionuclides and the vertical distribution of fine roots and fungal mycelia in soil. At least in the case of fungi, the use of transfer factors referring explicitly to the soil layer exploited by fungal mycelia seems to be the best approach for quantifying the uptake to radionuclides, balancing overall simplicity with mechanistic considerations of the dynamic processes [S37]. However, the concentrations of radionuclides in understorey vegetation, trees and fungal fruit bodies can be estimated roughly in a simplified manner using aggregated transfer factors. The ranges of aggregated transfer factors given in table 5 summarize the available observations.

Table 5. Typical ranges of aggregated transfer factors for ^{137}Cs from soil to vegetation and fungal fruit bodies in forest ecosystems [A8, B27, G7, I16, I17, K15, L7, Z1]

Data are given on a dry weight basis unless otherwise noted

Species or genus	*TF_{agg} (Bq/kg organism (d.m.) per Bq/m² soil)*
Fungal fruit bodies	
Agaricus	0.002–0.007
Amanita	0.008–5
Armillaria	0.001–0.2
Boletus	0.001–10
Cantharellus	0.01–2
Clitocybe	0.01–2
Collybia	0.03–0.3
Coprinus	0.004[a]
Cortinarius	0.02–10
Hydnum	3[a]
Hygrophorus	0.2–7
Laccaria	0.4–10
Lactarius	0.006–5
Leccinum	0.005–0.9
Lepista	0.002[a]
Lycoperdon	0.009–0.5
Macrolepiota	0.000 7–0.1
Paxillus	0.01–5
Ramaria	0.05–0.6
Rozites	0.08–10
Russula	0.04–5
Sarcodon	0.3–0.4
Suillus	0.02–2
Tuber	0.000 3–0.008[b]
Xerocomus	0.002–7

Species or genus	TF_{agg} *(Bq/kg organism (d.m.) per Bq/m² soil)*
Understorey vegetation	
Rubus chamaemorus (cloudberry), fruit	0.002–0.2
Vaccinium vitis-idaea (lingonberry), fruit	0.03–0.07
Vaccinium myrtillus (bilberry), fruit	0.02–0.1
Rubus idaeus (raspberry), fruit	0.001–0.004
Fragaria vesca (strawberry), fruit	0.004–0.01
Rubus fruticosus (blackberry), fruit	0.006–0.05
Green parts of understorey vegetation, including the stems of berry plants	0.001–1
Trees	
Fagus sp. (beech)	
Bole wood	0.001–0.002
Leaves	0.002–0.003
Picea sp. (spruce)	
Bole wood	0.000 3–0.002
Needles	0.000 6–0.02
Pinus sp. (pine)	
Bole wood	0.000 2–0.003
Needles	0.001–0.04
Quercus sp. (oak)	
Bole wood	0.002–0.004
Leaves	0.008–0.01
Betula sp. (birch)	
Bole wood	0.000 03–0.001
Leaves	0.000 3–0.04
Populus sp. (aspen)	
Bole wood	0.000 5–0.002
Leaves	0.008[a]
Alnus sp. (alder)	
Bole wood	0.001[a]
Leaves	0.008[a]

[a] Only a single value available.
[b] Data are given on a fresh weight basis and refer to the top 10 cm of soil.

76. Fungi are able to accumulate radiocaesium in their fruit bodies [G14, H8]. Some species exhibit activity levels that exceed those of green plants by more than one order of magnitude. On average, the radiocaesium levels in symbiotic fungi are higher than those in saprophytic species [R7, Y4, Y5].

77. Radionuclides in growing wood originate from two sources: the initial atmospheric deposits that enter the plant by foliar absorption, and root uptake from the soil. Their relative contributions depend on the type of tree (coniferous versus deciduous) and the age [B20, E7, G5, H9], the season at the time of deposition and the time elapsed after deposition, with root uptake being the dominant pathway for growing wood in the long term. Transfer factors or concentration ratios that are calculated on the basis of the total content of radionuclides in wood inevitably include both uptake processes and therefore are likely to overestimate root uptake (table 5) [G5].

6. Migration in soil

78. Vertical migration of radionuclides in the soil column is driven by various transport mechanisms, such as convection, dispersion, diffusion and bioturbation. The long-term consequences of downward migration differ considerably, however, depending on the dominant mechanism. For convective-driven migration, for example, the radionuclide input due to the Chernobyl accident moves down the soil as a marked peak and shows broadening with time as a result of dispersive mixing. Convective transport of radionuclides usually dominates in soils showing high hydraulic conductivities, e.g. sandy soils. For further discussion of the importance of downward migration of radionuclides in soil and forest litters, see section III and the references cited.

79. For diffusive transport, the concentration is always at a maximum at the surface with a close to exponential decrease with depth. For this type of transport, which is typical in

soils of low hydraulic conductivity, the bulk of the radionuclides deposited from the atmosphere thus remains within the rooting zone of plants.

80. Agricultural practices have a major impact on radionuclide behaviour. Depending on the intensity and type of soil cultivation, mechanical redistribution of radionuclides may occur. This causes, in arable soils, a rather uniform distribution of radionuclides in the tilled horizon. Fertilization shifts the ratio of radionuclide to nutrient concentrations in soil and soil solution and thus may influence plant root uptake of the radionuclides [E6].

81. Some investigations indicate [B21, S16] that element-independent transport mechanisms, such as the transport of radionuclides attached to clay particles or soil colloids, may play a relevant role in determining the migration rate of radionuclides in soil. Furthermore, the activity of soil animals that cause a turnover of soil, e.g. earthworms, cannot be neglected. The authors of references [B21, S16] suggest that a value of 100 years for the default residence half-time for the upper 25 cm layer is adequate for all elements with low mobility, such as radium, lead, uranium, plutonium and americium. Iodine under aerobic conditions is strongly bound to organic matter and therefore a residence half-time of 100 years can also be assumed [K7]. On the other hand, iodine can be released from soil to soil solution under anaerobic conditions, such as in a flooded paddy field [M25].

82. The situation with forest soil is more complex owing to the more pronounced soil horizons. Radionuclides deposited directly onto forest soil or washed from the canopy and understorey vegetation initially infiltrate the soil rather rapidly. They are therefore initially assigned to a labile pool. In the long term, they will become immobilized through fungal or microbial activity or by mineral constituents of the soil. The radionuclides in the non-labile pool may be available for root uptake, e.g. via symbiotic fungi, but are assumed not to be leached to deeper soil layers. The rate of downward migration is correspondingly reduced considerably over time, and, in the organic horizons, is determined mainly by the rates of decomposition of the organic material, and litter accumulation. Subsequently, downward migration of radionuclides is rather slow and partially offset by upward translocation by fungal mycelia and roots [R4]. Fungal and microbiological activity is likely to contribute substantially to the long-term retention of radionuclides, notably radiocaesium, in organic layers of forest soil. In this phase, radiocaesium is well mixed and almost equilibrated with stable caesium within the biologically connected compartments [Y6]. When radionuclides reach the mineral horizons of forest soil, essentially the same processes may occur as in arable soils, e.g. radiocaesium can be fixed by micaceous clay minerals.

7. Resuspension

83. Resuspension refers to the removal of deposited material from the ground to atmosphere as a result of wind, traffic, soil cultivation and other activities. Potentially, resuspension is a persistent source of radionuclides in air subsequent to their deposition on the ground. Furthermore, it may lead to redistribution of radionuclides and their deposition onto clean surfaces. Resuspension is influenced by a variety of factors, such as the time since deposition, meteorological conditions, surface characteristics and human activities. For biota, resuspension is of low importance. For animals living in the soil, it is not relevant. The contribution of resuspension to the activity concentration of radionuclides in plants in humid ecosystems usually is negligible compared to that of dry deposition and interception [G6, H10].

8. Transfer to animals

84. The transfer of radionuclides to animals is usually estimated using element-dependent concentration ratios or transfer factors. The transfer factor is defined either as the ratio of the activity concentration in an organism or tissue and the intake rate under equilibrium conditions, or as the ratio of the activity concentration in an organism or tissue and the deposition density (activity per unit area). It is only applicable to an intake of a radionuclide by adult animals that is constant over long periods. To account for time-dependent (dynamic) intakes, one or more biological half-lives are considered [M4].

85. In recent decades, many data have been accumulated on the transfer factors for domestic animals. They depend on animal mass, performance level, feeding regimes and feed components. However, these data are not generally applicable to estimating activity concentrations in biota, since they were determined in order to estimate activity concentrations in animal products for human food (such as meat, milk and eggs) while this annex is concerned with the estimation of activity concentrations in whole animals. Furthermore, the application of transfer factors presumes knowledge of the feed intake as well as the activity concentrations of the feed components. It has been demonstrated that highly contaminated feed components may determine the activity levels of game, even if consumed in low quantities. The seasonal peak activity concentration of ^{137}Cs in roe deer, for example, has been attributed to the ingestion of mushrooms [Z1]. Fungal fruit bodies can show radiocaesium levels exceeding those of green plants by one order of magnitude or more. Wild boar ingest deer truffle (*Elaphomyces granulatus*), a preferred "delicacy", which dominates the radiocaesium uptake, despite being only a few per cent of the boar's total diet [F14, P12]. However, the relevant data are not available for wild animals in general.

86. In most cases, the activity concentrations of radionuclides in game are calculated in a simplified manner using aggregated transfer factors. This transfer factor neither takes into account the time-dependent intake rates nor can reproduce the time-dependent activity concentrations in game. Values for aggregated transfer factors for different species are compiled in table 6.

Table 6. Aggregated transfer factors (soil-to-game) for ^{137}Cs [A9, I16, J3, K8, S19, Z1]
Data are given on a fresh mass basis unless otherwise noted

Species	*TF_{agg} (Bq/kg organism (dry mass) per Bq/m² soil (dry mass))*	
	Default value	*Range of literature data*
Alces alces (moose)	0.02	0.006–0.03
Capreolus capreolus (roe deer)	0.05	0.001–0.2
Cervus elaphus (red deer)	0.03	0.02–0.04
Lepus arcticus (arctic hare)	0.03	0.009–0.1
Lepus capensis (brown hare)	0.004	0.002–0.05
Lynx lynx (lynx)	0.3	0.01–10[a]
Game except roe deer	0.02	

[a] Data are given on a dry weight basis.

87. Table 7 summarizes the equilibrium concentration ratios for the reference organisms considered. The values are "order-of-magnitude" estimates based on the compilation in reference [F4]. Some of the original values were given as aggregated transfer factors and have been converted to concentration ratios. At least in temperate environments, concentration ratios are higher in forest and semi-natural ecosystems than in agricultural systems, because of their often lower nutrient supply and pH values. Furthermore, the high content of organic matter in forests is accompanied by high concentrations of fulvic and humic acids, which act as complexing agents and increase the mobility of cationic radionuclides in soil.

88. The nominal values of transfer factors provided in table 7 have been suggested for use [E10, F4], in the absence of site-specific information, to estimate the exposure rates for biota after the release of radionuclides to atmosphere and their subsequent transfer to soil. As such, these transfer factors were intended to be applied for screening purposes to obtain an order of magnitude estimate, but they may not be appropriate for application to specific sites.

Table 7. Nominal values of transfer factors for reference organisms (adapted from [E10, F4])

Element	*Transfer factors (Bq/kg (fresh weight) per Bq/kg soil)*							
	Earthworm	*Rat*	*Deer*	*Duck*	*Frog*	*Bee*	*Grass*	*Pine tree*
H	150	150	150	150	150	150	150	150
Cl	0.2	7	7	7	7	0.3	20	1
Sr	0.01	2	2	0.6	1	0.06	0.2	0.5
Tc	0.4	0.4	0.4	0.4	0.4	0.4	20	0.3
I	0.2	0.4	0.4	0.4	0.4	0.3	0.1	0.1
Cs	0.09	3	3	0.8	0.6	0.06	0.7	0.2
Np	0.1	0.04	0.04	0.04	0.04	0.1	0.02	0.3
Pu	0.03	0.02	0.02	0.02	0.02	0.06	0.01	0.03
Am	0.1	0.04	0.04	0.04	0.04	0.1	0.005	0.000 1
Pb	0.03	0.04	0.04	0.06	0.1	0.06	0.07	0.08
Ra	0.09	0.03	0.03	0.04	0.04	0.04	0.04	0.000 7
Th	0.009	0.000 1	0.000 1	0.000 4	0.000 4	0.009	0.04	0.001
U	0.009	0.000 1	0.000 1	0.000 5	0.000 5	0.009	0.02	0.007

C. Transfer to freshwater organisms

89. Radionuclides can enter water bodies as a result of discharges to the aquatic environment (e.g. directly from a nuclear facility), by deposition of airborne radioactive material onto the water surface and by run-off of material deposited onto soil. For a point source of emission into a swiftly flowing stream, the flow rate of the stream can be divided by the flow rate of the effluent discharge to obtain the dilution factor. A certain mixing distance must be assumed, which could vary from a few tens of metres for a small stream to a few kilometres for a large river. Beyond the

mixing distance, a uniform concentration of the radionuclide in water can be assumed. Suspended material may be deposited as sediment. The deposited material may become locked in the sediments and, over time, migrate to deeper sediments or be redissolved by physical and biological processes and re-enter the water column. Dissolved or finely suspended material may be transported over large distances, being progressively diluted by water from other streams and rivers, eventually reaching the oceans.

90. The movement of radionuclides in rivers is often modelled using the diffusion–transport equation and the behaviour of radionuclides in the "water column–river bed sediment" system is often assessed using compartment models [M23]. At present, although the structures of the models have not been subjected to significant revisions, the scope of the transfers modelled (physical, chemical and biological) and of the associated radionuclide specific parameters has been considerably enlarged. For instance, the previous state-of-the-art publication of the IAEA, "Handbook of parameter values for the prediction of radionuclide transfer in temperate environments" [I16], listed solely values of water–sediment partition coefficients and concentration factors for edible portions of fish. The most recent version also incorporated equations and parameters for representing transfer by wash-off from watersheds of deposited radionuclides, interaction between liquid and solid phases, migration to and from sediments, and transfers to freshwater biota [I14].

91. The mixing of radionuclides discharged into a lake or pond is much slower than is the case for rivers. As a first approximation, a uniform radionuclide concentration throughout the pond could be assumed, with a dilution factor equal to the pond outflow rate divided by the effluent input rate. In a large lake or coastal environment, a uniform concentration would never be reached. Plume models have been developed for lake-shore environments analogous to atmospheric transport models. The lake-shore environment is often complicated by thermal layering within the water column, which impedes vertical mixing. Moreover, removal of material from the water column via sedimentation is an important long-term process which results in an approximately exponential decline with time of the radionuclide concentrations present in the water column.

92. Sedimentation and attachment to suspended particulates are the main processes influencing the residence times of radionuclides in freshwater. Fractions of dissolved and of particle-bound radionuclides are usually determined by the distribution coefficient, K_d, which is defined as the ratio of the radionuclide concentration in water and the concentration of the radionuclide attached to particulate matter, under equilibrium conditions. Values of K_d are element-dependent. Low K_d values and concentrations of suspended matter indicate high dissolved fractions, whereas high K_d values and suspended load values indicate a considerable sorption of radionuclides by particles and favour sedimentation. Once deposited, radionuclides may migrate down within the sediment or may become involved in resuspension processes. These processes may create additional sources or sinks with potential impact on the long-term behaviour. The distribution coefficients for various elements in freshwater are given in table 8.

Table 8. Distribution coefficients K_d in freshwater ecosystems [I14]

Element	*K_d (m³/kg)*	
	Geometric mean	*Geometric standard deviation*
Be	42	3.6
Mn	130	12
Co	43	9.5
Sr	0.18	4.6
Ru	32	1.9
Ag	85	2.3
Sb	5	3.8
I	4.4	14
Cs	8.5	6.7
Ba	2	3.6
Ce	220	2.8
Th	180	21
Ra	7.4	3.1
Pu	240	6.6
Am	850	3.7

93. Aquatic organisms may be directly irradiated by radionuclides present in their habitats (e.g. water, sediment). They may also take up radionuclides from water and/or the food chain and incorporate them into their tissues. External irradiation of most aquatic organisms, with the exception of burrowing invertebrates and benthic organisms, is limited by the shielding provided by the surrounding water or sediment.

94. Considerable attention has been focused on fish because they are at a higher trophic level in aquatic food chains and serve as food for humans and predators. Polikarpov [P2] has given concentration ratios, CR, (CR here is the ratio of the activity concentration in fish expressed in units of Bq/kg and that in water expressed in units of Bq/L, under equilibrium conditions) for ^{137}Cs ranging from 500 to 9,500 L/kg for freshwater fish, compared to values of 3 to 25 L/kg for marine fish. The lower values for marine fish were thought to be as a result of the competition for uptake from potassium and other cations. Freshwater amphibians can also show high values of CR (1,000 to 8,000 L/kg) in the aqueous environment.

95. Table 9 gives values of CR for ^{137}Cs in fish in Canadian lakes in the Northwest Territories [L5] and for the upper Great Lakes [T15]. High trophic level fish such as trout, pike and cisco show an especially high accumulation of radiocaesium.

Table 9. Concentration ratios for ^{137}Cs in freshwater fish

Species	*Concentration ratio (L/kg)*	
	NWT Lakes [L5]	*Great Lakes [T15]*
Burbot	800	
Lake whitefish	400–1 000	
Round whitefish	1 000–1 800	
Sucker	700	1 500–2 500
Chub		1 900
Alewife		1 800–2 300
Bullhead		2 300
Cisco	1 600–5 000	
Pike		2 500–5 500
Lake trout	3 000–6 000	6 100

96. Swanson [S20] has summarized concentration ratios for water to fish tissues for the naturally occurring radionuclides of uranium, ^{226}Ra, ^{210}Pb, and ^{228}Th (table 10).

Table 10. Concentration ratios for natural radionuclides in freshwater fish [S20]

Element/ radionuclide	*Concentration ratio (L/kg)*					
	Bone	*Flesh*	*Liver*	*Kidney*	*Gonad*	*Gut*
U	20–800	0.1–25	<0.04–0.5	0.1–0.5	0.01–0.35	0.05–0.5
^{226}Ra	35–1 800	1–60	1–45	3–30	5–115	7–45
^{210}Pb	100–2 500	4–100	3–420	6–780	10–150	11–206
^{228}Th	15–160	4–32	4–36	5–46	13–50	23–50

D. Transfer of radionuclides to marine organisms

97. The main processes that modify the activity concentrations of radionuclides in marine water are *(a)* dilution due to convective and dispersive mixing during transport, driven by local, regional and global currents, *(b)* sedimentation after attachment to suspended particles and *(c)* radioactive decay.

98. For a given continuous discharge rate into a specific section of the marine system, the steady-state concentration of a dissolved radionuclide in water, C_w (Bq/m³), can be calculated according to:

$$C_w = \frac{A}{V \cdot (\tau^{-1} + \lambda_r)} \cdot \frac{1}{1 + K_d \cdot S} \qquad (2)$$

where A is the activity of the radionuclide discharged per unit time to a specific part of the sea (Bq/a), V is the volume of this part (m³), τ is the mean residence time *(a)*, λ_r is the radioactive decay constant (a⁻¹), K_d is the distribution coefficient (m³/kg), and S is the concentration of suspended particles (kg/m). The steady-state activity concentration of the radionuclide in suspended particles, C_s (Bq/kg), is then:

$$C_s = \frac{A}{V \cdot (\tau^{-1} + \lambda_r)} \cdot \frac{K_d}{1 + K_d \cdot S} \qquad (3)$$

The distribution coefficients for a number of elements in marine waters are summarized in table 11.

Table 11. Distribution coefficients K_d for open ocean and ocean margins [I20]

Element	*K_d (m³/kg)*	
	Open ocean	*Ocean margins*
H	0.001	0.001
Cl	0.001	0.000 3
Sr	0.2	0.008
Tc	0.1	0.1
I	0.2	0.07
Cs	2	4
Pb	1×10^4	1×10^2
Ra	4	2
Th	5×10^3	3×10^3
U	0.2	1
Np	1×10^2	1
Pu	2×10^3	1×10^2
Am	2×10^3	2×10^3

99. A value of 3 years was given in reference [U3] for the mean residence time, t, in a specified part of the marine system, for all radionuclides in coastal waters with the exception of ^{239}Pu, for which a value of 3.5 years was assumed. These values took account of radionuclide losses from water to sediment. From simulations of the transport of radionuclides discharged from the reprocessing plants at Sellafield and La Hague through the North Atlantic and its marginal seas, the mean residence times given in table 12 were estimated using the North Atlantic–Arctic Ocean Sea Ice Model (NAOSIM) [I21].

Table 12. Residence times in different parts of the North Atlantic according to the NAOSIM model

Part of ocean	*Volume (km³)*	*Mean residence time (a)*
North Sea	41 000	2.5 ± 0.36
Norwegian Sea	59 000	0.37 ± 0.11
Barents Sea	220 000	2.4 ± 0.24
Kara Sea	38 000	4.5 ± 1.2
Central Nordic Seas	44 000	0.52 ± 0.18

100. As for freshwater aquatic biota, activity concentrations of radionuclides in marine biota can be estimated using a concentration ratio approach. Concentration ratios for various elements in marine biota are compiled in table 13. For most elements, these data are based on concentrations in muscle (fish) and soft tissue (crustaceans). For the bone seeking elements such as strontium, however, the entries in table 13 are based on whole body concentrations.

Table 13. Concentration ratios for marine biota [I20]

Element	*Concentration factors (L/kg fresh weight)*		
	Fish	*Macroalgae*	*Crustaceans*
H	1	1	1
Cl	0.06	0.05	0.06
Sr	3	10	10
Tc	80	30 000	1 000
I	9	10 000	100
Cs	100	50	30
Np	1	50	100
Pu	100	4 000	200
Am	100	8 000	400
Pb	200	1 000	9 000
Ra	100	100	100
Th	600	200	1 000
U	1	100	10

E. Evaluating doses to biota

1. Fraction of radiation absorbed by organism

101. Radionuclides distributed in the environment lead to external exposure of an organism living in or close to a medium that contains radionuclides. The external exposure of biota is the result of complex and non-linear interactions of various factors:

- The geometrical relation between the source of the radiation and the target;
- The activity levels of the radionuclides in the environment;
- The materials in the environment and their shielding properties;
- The radionuclide-specific decay properties characterized by the radiation type, the energies emitted and their emission probabilities; and
- The habitat and size of the organism.

102. The geometric relationship between the radiation source and the exposed organism is an important factor in relation to the absorbed dose rate incurred. The intensity of the radiation field around a source decreases with distance and is influenced by the material between the radiation source and the target. The number of possible source target configurations is infinite; therefore, a number of limited and representative situations need to be selected for detailed consideration.

103. The exposure due to radionuclides incorporated into the organism is determined by the activity concentrations in the organism, the size of the organism, and the type and the energy of the emitted radiation. A key quantity for estimating internal doses is the absorbed fraction of energy, $\phi(E)$, which is defined as the fraction of energy emitted by a radiation source that is absorbed within the target tissue, organ or organism. In the simplest case, the organism is assumed to be in an infinite homogeneous medium and to have a uniform activity concentration throughout its body. The densities of the medium and the organism's body are assumed to be identical. Under these conditions, both internal (D_{int}) and external (D_{ext}) dose conversion coefficients (DCCs; the DCC is defined as either the absorbed dose or the absorbed dose rate, according to the circumstances, per unit activity concentration of the relevant radionuclide in the organism or medium) for monoenergetic radiation can be expressed as a function of the absorbed fraction [N1, V2]:

$$D_{int} = E \cdot \phi(E) \quad \text{and} \quad D_{ext} = E \cdot \left(1 - \phi(E)\right) \qquad (4)$$

104. Absorbed fractions for photon and electron sources uniformly distributed in soft-tissue spherical bodies immersed in an infinite water medium have been systematically calculated by Monte Carlo simulation [U17]. The calculations covered a particle energy range of 10 keV to 5 MeV, a range for the mass of the body from 10^{-6} to 10^{3} kg, and shapes from spheres to ellipsoids with varying degrees of non-sphericity. Figures III and IV show, respectively, the results for electrons and photons.

Figure III. Absorbed fraction, $\phi(E)$, for electrons of different energy uniformly distributed in spheres of different mass in a water medium

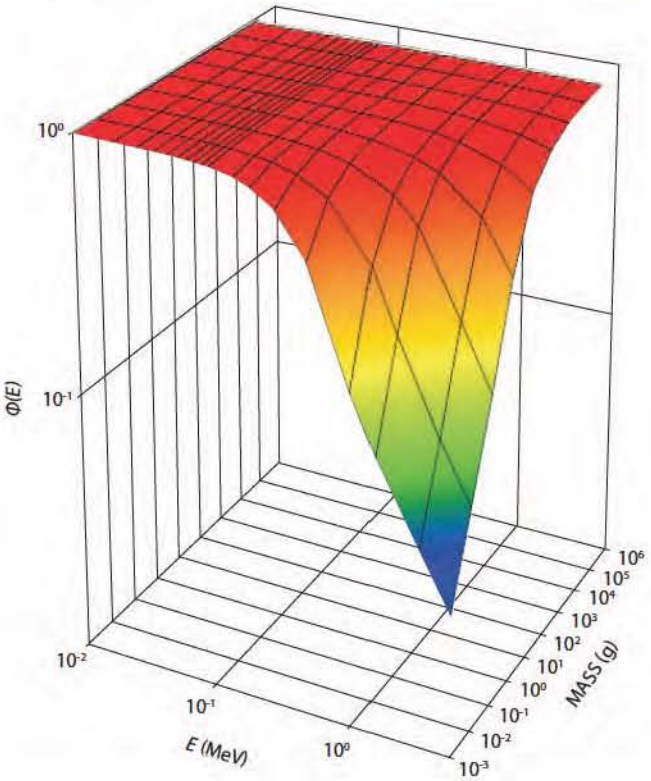
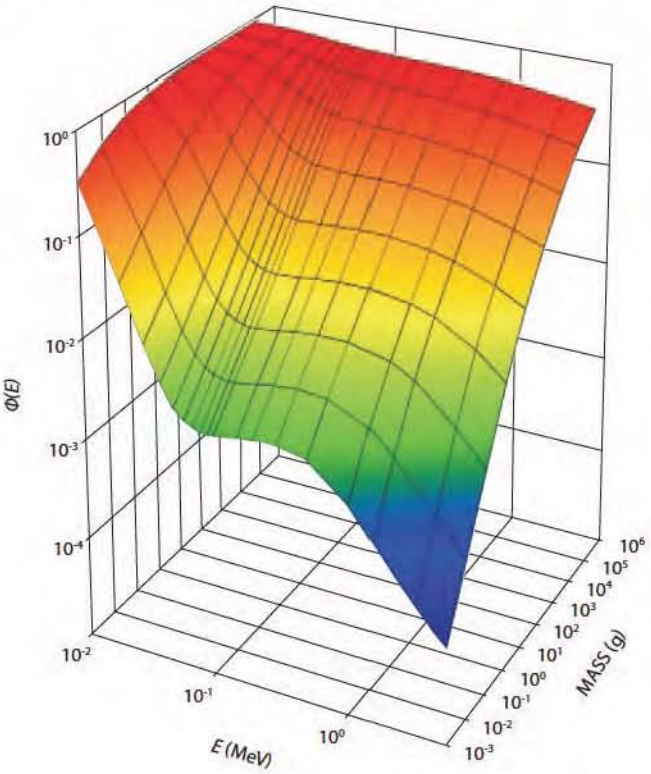


Figure IV. Absorbed fraction, $\phi(E)$, for photons of different energy uniformly distributed in spheres of different mass in a water medium



105. For electron energies below 100 keV, the absorbed fraction is close to unity, even for very small organisms. The mean free path of electrons in living tissue increases from 160 μm for 100 keV electrons to 5 mm for 1 MeV electrons. Thus, even above 100 keV, the absorbed fraction is close to unity if the diameter of the target is much greater than the range of the electron. Only for very small targets and high energies does the absorbed fraction become considerably smaller than 0.5.

106. The mean free path of photons is considerably longer than that of electrons. The absorbed fractions cover a range from nearly unity for low photon energies and large organisms to less than 0.0001 for small organisms and high photon energies. Absorption is a non-linear function of target size and energy. The main processes causing absorption of photon energy are the Compton effect, the photoelectric effect and pair production; their contributions to absorption depend on the energy of the emitted photons. As a result, the absorbed fraction of photons in the energy range from 20 to 100 keV decreases by a factor of 10–15 for small organisms, but is relatively constant for photons with energies between 100 keV and 1 MeV. Beyond energies of 1 MeV, the absorbed fraction decreases steeply with energy.

107. The range of alpha particles in living tissue is very short, increasing from 16–130 μm within the energy range of 3–10 MeV. Therefore, with the exception of bacteria, it is assumed for all organisms that all the energy emitted is absorbed. Since the dimensions of bacteria are well below the range of alpha particles, the absorbed fraction is assumed to be zero.

108. Re-scaling factors have been derived from the computed absorbed fractions for spheres to determine the dose coefficients for ellipsoidal shaped organisms, using the mass and proportions of the organism. The relationship between the re-scaling factors and the non-sphericity parameter of the organism's body are described analytically in reference [U17]. Owing to the short range of alpha particles, the internal exposure due to incorporated alpha emitters is independent of the shape of the organism.

109. The approach was also applied to the calculation of the absorbed fractions for non-aquatic animals and their internal exposures. With the use of the absorbed fractions for spheres and the suggested re-scaling and interpolation techniques, a set of internal DCCs has been calculated for all reference animals and plants [U17].

110. The estimation of external exposures of terrestrial reference animals and plants is more complex than that of biota in the aquatic environment. The intrinsically different density and composition of soil, air and organic matter cannot, in general, be adequately taken into account by the application of analytical solutions. Dosimetric models for estimating external doses to biota in the terrestrial environment were developed within the FASSET project [F4, T10]. A key factor for determining external exposure is the geometric

relationship between the radiation source and the exposed organism. A number of limited and representative exposure situations were selected for detailed consideration.

111. Simple three-dimensional phantoms, i.e. ellipsoids and cylinders, were defined as model geometric equivalents of reference organisms based on their average mass and size characteristics. The dimensions ranged from a millimetre to a metre and the respective masses range from 0.2 g to 550 kg. The ellipsoids represented organisms such as woodlouse, earthworm, mouse, mole, snake, fox, deer and cattle. Details of the assumed exposure conditions are given in reference [T10]. The fur and the outer layers of skin consist of non-active tissue, and therefore shield the living organism.

112. Herbaceous vegetation, shrubs and trees were considered as reference plants. Exposure of the meristem and buds was calculated because these organs are characterized by very intensive cell division, which may make them highly radiosensitive.

113. In order to take account of the distribution of radionuclides in the canopy, a distinction was made between alpha, beta and gamma radiation because of their different ranges. For gamma radiation, the whole canopy was considered to have a homogeneous activity concentration. For high-energy beta radiation, the irradiation of the target was also assumed to result from a canopy with a homogeneous activity concentration. However, owing to the much shorter range of alpha and low-energy beta radiation, the irradiation resulting from external deposits on, or internal activity of, the target organ had to be considered explicitly. Because of the very short range of alpha particles in air, only the exposure due to the external deposits on, or internal exposure of, the target needed to be taken into account [T10].

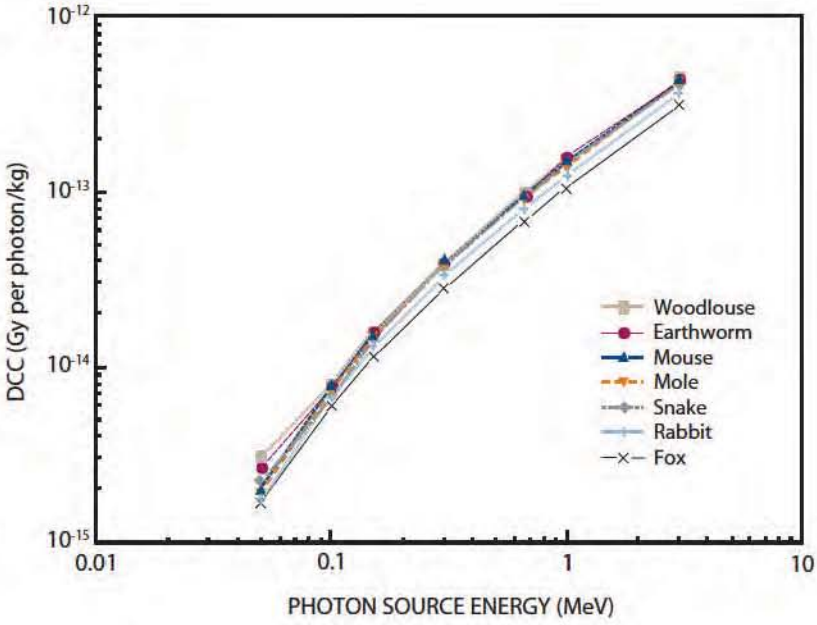
114. The elemental composition and density of the materials involved have an important impact on the radiation transport calculation. All organisms were assumed to be composed of skeletal muscle alone with the characteristics/parameters given in reference [I15]. The DCCs were derived using Monte-Carlo techniques; all relevant processes of radiation transport and interaction with matter were included. For electrons, a thick-target bremsstrahlung model was used instead of an electron-transport simulation. For the calculation of DCCs for a species in the soil, a volume source with uniform activity concentration was assumed. For the calculation of DCCs for a species on the ground, a planar radiation source on top of the soil with a surface roughness of 3 mm and a volume source with a depth of 10 cm were assumed. Calculations were made for monoenergetic gamma energies of 50 keV, 300 keV, 662 keV, 1 MeV and 3 MeV. Data for other energies were obtained by interpolation.

2. Principal relationships for internal and external exposure

(a) External exposure

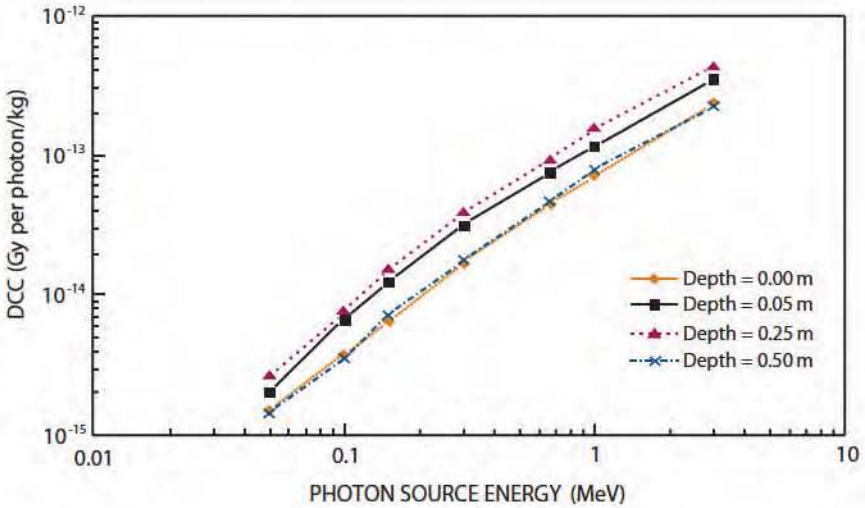
115. Although the simulations cover only a limited number of possible exposure conditions, they allow the relationships between organism size, radiation energy and habitat to be deduced. The DCC (Gy per photon per kg) increases in proportion to the photon energy as illustrated in figure V for a volumetric source with a thickness of 0.5 m and target organisms that live at a depth of 0.25 m. Whereas the DCCs vary by a factor of 200 between photon energies of 50 keV to 3 MeV, the variation between the organisms does not exceed a factor of 2, even for low-energy photons (for high-energy photons, the difference is a factor of only 1.5).

Figure V. Dose conversion coefficients for various soil organisms at a soil depth of 25 cm, for monoenergetic photons from a uniformly distributed source in the upper 50 cm of soil (soil density: 1,600 kg/m³) [F4]



116. The DCC (Gy per photon per kg) for an earthworm as a function of soil depth for monoenergetic photons is shown in figure VI. The upper 50 cm of the soil was assumed to have a homogeneous activity concentration. The maximum DCC was found to be at a depth of 25 cm and the lowest, at depths of 0 cm and 50 cm. The maximum DCC is a factor of 2 higher than the lowest.

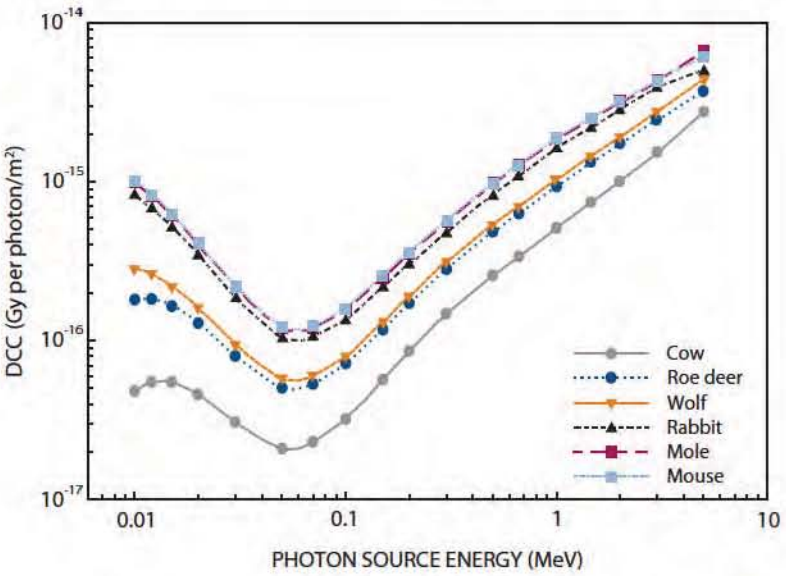
Figure VI. Dose conversion coefficients for an earthworm at various depths in soil, for monoenergetic photons from a uniformly distributed source in the upper 50 cm of the soil (soil density: 1,600 kg/m³) [F4]



117. The DCC at a depth of 5 cm is only about 20% lower than the maximum. This is because of the relatively short mean free path of photons in soil, which is about 0.2, 2 and 10 cm for photon energies of 20 keV, 100 keV and 3 MeV, respectively. Thus, an organism in soil would be irradiated by photons originating within a surrounding shell of, at most, 10 cm radius.

118. The DCCs (Gy per photon per m²) for different reference organisms for a planar source on the soil surface are given in figure VII. The DCCs decrease as the photon energy increases from 10 to 100 keV by a factor of about 5 for small animals and 2 for large animals. Above 100 keV, the DCCs gradually increase by approximately two orders of magnitude; the DCCs for small animals are greater than those for large animals owing to the more effective self-shielding in large organisms. Such differences are more pronounced at low energies; for example, the difference between the mouse and the cow is a factor of about 6 for 50 keV photons, whereas it is a factor of 3 for 3 MeV photons.

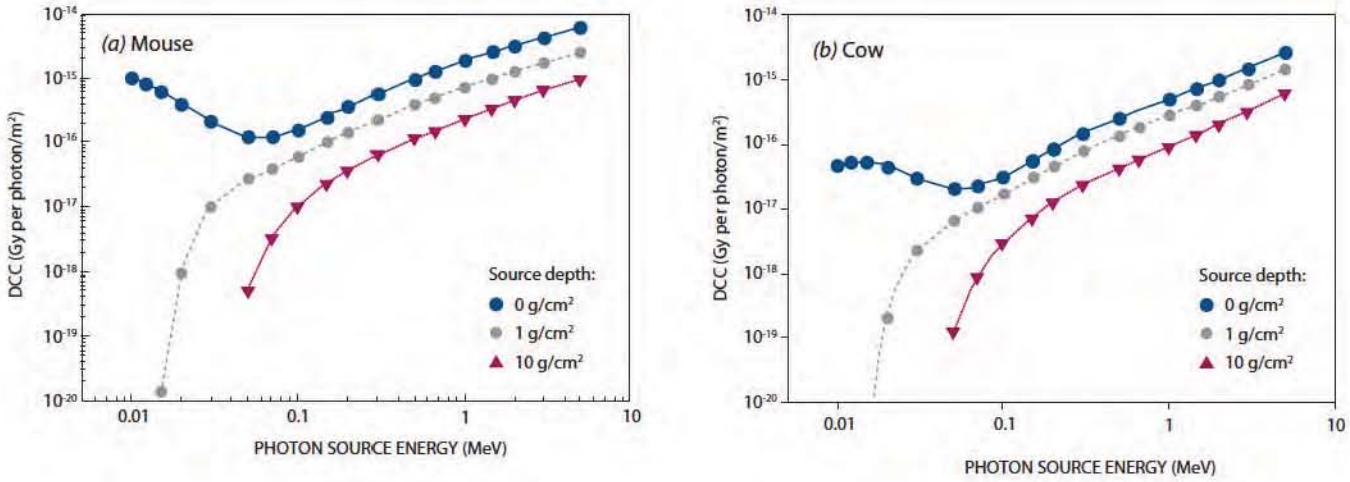
Figure VII. Dose conversion coefficients as a function of the source energy for various reference organisms for a planar source on top of the soil [F4]



119. The DCCs for different animals as a function of the depth of a planar source in the soil are summarized in figure VIII. The results show that the DCCs for low-energy photons for animals living on soil are low. Relatively shallow depths of soil over the planar source suffice to attenuate the photons completely.

Figure VIII. Dose conversion coefficients as a function of the source energy and depth of a planar source in the soil for *(a)* the mouse and *(b)* the cow living on the soil

The source depth quantifies the amount of soil by which the photon source is covered (e.g. the source depth of 10 g/cm² for soil densities of 1.0 and 1.6 g/cm² are equivalent to a depth of the source in the soil of 10 and 6.25 cm, respectively) [F4]



120. The data indicate that the relationship between the DCCs, the size and habitat of the organism and the energy and type of the radiation is complex. Nevertheless, these data provide an appropriate basis for deriving data, either by interpolation or by extrapolation, for other exposure conditions that were not explicitly considered. They were used to derive radionuclide-specific DCCs (μGy/h per Bq/kg) for internal and external exposure of a number of reference organisms, taking into account the type of radiation as well as the energy and intensity of the emission, as specified by the ICRP [I13]. Table 14 summarizes the DCCs (μGy/h per Bq/kg) for external exposure. The data are provided according to the habitat of organisms considered. Animals living in soil were assumed to be at a depth of 25 cm in a soil layer that is homogeneously contaminated by radionuclides to a depth of 50 cm. Above ground organisms were assumed to be irradiated by a source homogeneously distributed in the soil layer to a depth of 10 cm. For the terrestrial organisms, only the contribution of photons was included, whereas for aquatic organisms, exposure due to electrons (including bremsstrahlung) was also implicitly taken into account. This has the effect of causing the DCCs for ^{3}H, ^{90}Sr and ^{135}Cs to appear to be inconsistent: the DCCs for ^{3}H and ^{135}Cs for terrestrial organisms are zero, whereas the values for aquatic organisms are very small. Aquatic organisms are in direct contact with the contaminated medium, whereas electrons emitted from soil are attenuated by the surface roughness of the soil, the air and the fur of terrestrial organisms. So, this apparent inconsistency is of no significant practical consequence.

(b) Internal exposure

121. The DCCs (μGy/h per Bq/kg) for internal exposure are provided in table 15 [U17]. The values are given in terms of weighted absorbed dose rate per unit activity concentration in the organism, assuming homogeneous distribution of the radionuclides. The DCCs have been weighted to take account of the different RBEs of the different qualities of radiation; a factor of 10 to reflect the RBE has been used for alpha radiation and a factor 1 to reflect that for gamma and beta radiation including that from tritium (see the next subsection).

Table 14. Dose conversion coefficients for external exposure of reference organisms [E10, T10, U17]

Radionuclide	*Absorbed dose rates per activity concentration (μGy/h per Bq/kg)*											
	In soil[a]		*On soil*[b]						*In water*[c]			
	Earthworm	*Rat*	*Deer*	*Duck*	*Frog*	*Bee*	*Grass*	*Pine tree*	*Pelagic fish*	*Benthic fish*	*Brown seaweed*	*Crab*
^{3}H	0	0	0	0	0	0	0	0	7.9×10^{-12}	7.8×10^{-12}	8.2×10^{-11}	1.0×10^{-11}
^{90}Sr	1.5×10^{-10}	1.2×10^{-10}	4.6×10^{-12}	1.5×10^{-11}	1.6×10^{-11}	1.6×10^{-11}	1.6×10^{-11}	1.6×10^{-11}	2.2×10^{-5}	4.9×10^{-5}	2.0×10^{-4}	2.2×10^{-5}
^{99}Tc	0	0	0	0	0	0	0	0	1.2×10^{-7}	1.2×10^{-7}	1.5×10^{-6}	1.4×10^{-7}
^{129}I	3.5×10^{-6}	3.0×10^{-6}	4.0×10^{-7}	1.1×10^{-6}	1.1×10^{-6}	1.1×10^{-6}	1.1×10^{-6}	1.1×10^{-6}	7.7×10^{-6}	9.2×10^{-6}	1.3×10^{-5}	7.9×10^{-6}
^{131}I	1.9×10^{-4}	1.8×10^{-4}	3.7×10^{-5}	7.1×10^{-5}	7.7×10^{-5}	7.8×10^{-5}	7.8×10^{-5}	7.8×10^{-5}	1.9×10^{-4}	2.0×10^{-4}	2.3×10^{-4}	1.9×10^{-4}
^{134}Cs	8.3×10^{-4}	7.8×10^{-4}	1.6×10^{-4}	2.9×10^{-4}	3.2×10^{-4}	3.2×10^{-4}	3.2×10^{-4}	3.2×10^{-4}	7.6×10^{-4}	8.0×10^{-4}	8.9×10^{-4}	7.6×10^{-4}
^{135}Cs	0	0	0	0	0	0	0	0	4.5×10^{-8}	4.5×10^{-8}	4.3×10^{-7}	5.2×10^{-8}
^{137}Cs	3.0×10^{-4}	2.8×10^{-4}	5.6×10^{-5}	1.1×10^{-4}	1.1×10^{-4}	1.2×10^{-4}	1.1×10^{-4}	1.1×10^{-4}	2.7×10^{-4}	2.9×10^{-4}	3.4×10^{-4}	2.8×10^{-4}
^{210}Pb	6.0×10^{-7}	5.2×10^{-7}	7.7×10^{-8}	2.6×10^{-7}	2.8×10^{-7}	2.9×10^{-7}	2.9×10^{-7}	2.9×10^{-7}	2.7×10^{-6}	4.7×10^{-6}	4.7×10^{-5}	3.0×10^{-6}
^{226}Ra	9.0×10^{-4}	8.5×10^{-4}	1.8×10^{-4}	3.2×10^{-4}	3.4×10^{-4}	3.5×10^{-4}	3.4×10^{-4}	3.4×10^{-4}	9.1×10^{-4}	9.6×10^{-4}	1.1×10^{-3}	9.1×10^{-4}
^{232}Th	1.4×10^{-7}	1.2×10^{-7}	1.3×10^{-8}	3.9×10^{-8}	4.3×10^{-8}	4.4×10^{-8}	4.3×10^{-8}	4.3×10^{-8}	1.5×10^{-7}	1.8×10^{-7}	5.0×10^{-7}	1.6×10^{-7}
^{238}U	1.2×10^{-7}	1.0×10^{-7}	1.0×10^{-8}	4.3×10^{-8}	4.8×10^{-8}	5.0×10^{-8}	4.9×10^{-8}	4.9×10^{-8}	1.0×10^{-7}	1.3×10^{-7}	5.1×10^{-7}	1.1×10^{-7}
^{237}Np	7.6×10^{-6}	7.0×10^{-6}	1.4×10^{-6}	3.3×10^{-6}	3.6×10^{-6}	3.6×10^{-6}	3.6×10^{-6}	3.6×10^{-6}	1.2×10^{-5}	1.3×10^{-5}	1.8×10^{-5}	1.3×10^{-5}
^{238}Pu	8.5×10^{-8}	7.2×10^{-8}	9.5×10^{-9}	3.0×10^{-8}	3.3×10^{-8}	3.3×10^{-8}	3.3×10^{-8}	3.3×10^{-8}	8.2×10^{-8}	1.0×10^{-7}	3.0×10^{-7}	8.7×10^{-8}
^{240}Pu	1.6×10^{-7}	1.3×10^{-7}	1.4×10^{-8}	5.4×10^{-8}	6.0×10^{-8}	6.2×10^{-8}	6.1×10^{-8}	6.1×10^{-8}	1.4×10^{-7}	1.9×10^{-7}	6.8×10^{-7}	1.6×10^{-7}
^{241}Am	6.1×10^{-6}	5.5×10^{-6}	9.2×10^{-7}	2.4×10^{-6}	2.6×10^{-6}	2.6×10^{-6}	2.6×10^{-6}	2.6×10^{-6}	1.1×10^{-5}	1.2×10^{-5}	1.7×10^{-5}	1.1×10^{-5}

[a] Organisms are assumed to live at 25 cm depth of a soil with radionuclides distributed homogeneously to a depth of 50 cm.
[b] Organisms are assumed to live on a soil layer with radionuclides distributed homogenously to a depth of 10 cm.
[c] Organisms are assumed to be immersed in water.

Table 15. Weighted dose conversion coefficients for internal exposure of reference organisms [T10, U20]

Radionuclide	*Weighted absorbed dose rates per activity concentration (μGy/h per Bq/kg)*[a,b]											
	Earthworm	*Rat*	*Deer*	*Duck*	*Frog*	*Bee*	*Grass*	*Pine tree*	*Pelagic fish*	*Benthic fish*	*Brown seaweed*	*Crab*
^{3}H	3.3×10^{-6}											
^{36}Cl	1.5×10^{-4}	1.6×10^{-4}	1.6×10^{-4}	1.6×10^{-4}	1.6×10^{-4}	1.5×10^{-4}	1.5×10^{-4}	1.6×10^{-4}	1.6×10^{-4}	1.6×10^{-4}	1.4×10^{-4}	1.6×10^{-4}
^{90}Sr	5.3×10^{-4}	6.2×10^{-4}	6.5×10^{-4}	6.3×10^{-4}	5.9×10^{-4}	4.4×10^{-4}	5.1×10^{-4}	6.5×10^{-4}	6.3×10^{-4}	6.0×10^{-4}	4.5×10^{-4}	6.3×10^{-4}
^{99}Tc	5.8×10^{-5}	5.8×10^{-5}	5.8×10^{-5}	5.8×10^{-5}	5.8×10^{-5}	5.7×10^{-5}	5.8×10^{-5}	5.8×10^{-5}	5.8×10^{-5}	5.8×10^{-5}	5.7×10^{-5}	5.8×10^{-5}
^{129}I	3.8×10^{-5}	4.2×10^{-4}	5.0×10^{-5}	4.4×10^{-5}	3.9×10^{-5}	3.7×10^{-5}	3.8×10^{-5}	5.0×10^{-5}	4.3×10^{-5}	4.2×10^{-5}	3.8×10^{-5}	4.3×10^{-5}
^{131}I	1.1×10^{-4}	1.4×10^{-4}	2.6×10^{-4}	1.5×10^{-4}	1.2×10^{-4}	1.1×10^{-4}	1.1×10^{-4}	2.5×10^{-4}	1.4×10^{-4}	1.3×10^{-4}	1.0×10^{-4}	1.4×10^{-4}
^{134}Cs	1.1×10^{-4}	1.9×10^{-4}	7.1×10^{-4}	2.5×10^{-4}	1.4×10^{-4}	9.9×10^{-5}	1.1×10^{-4}	6.5×10^{-4}	2.3×10^{-4}	1.9×10^{-4}	9.7×10^{-5}	2.3×10^{-4}
^{135}Cs	3.9×10^{-5}	3.9×10^{-5}	3.9×10^{-5}	3.9×10^{-5}	3.9×10^{-5}	3.8×10^{-5}	3.9×10^{-5}	3.9×10^{-5}	3.9×10^{-5}	3.9×10^{-5}	3.8×10^{-5}	3.9×10^{-5}
^{137}Cs	1.5×10^{-4}	1.8×10^{-4}	3.7×10^{-4}	2.0×10^{-4}	1.6×10^{-4}	1.4×10^{-4}	1.4×10^{-4}	3.5×10^{-4}	1.9×10^{-4}	1.8×10^{-4}	1.3×10^{-4}	1.9×10^{-4}
^{210}Pb	2.3×10^{-4}	2.4×10^{-4}	2.5×10^{-4}	2.5×10^{-4}	2.4×10^{-4}	2.2×10^{-4}	2.3×10^{-4}	2.5×10^{-4}	2.5×10^{-4}	2.4×10^{-4}	2.0×10^{-4}	2.5×10^{-4}
^{226}Ra	1.3×10^{-1}	1.3×10^{-1}	1.4×10^{-1}	1.4×10^{-1}	1.3×10^{-1}	1.4×10^{-1}	1.3×10^{-1}	1.4×10^{-1}	1.4×10^{-1}	1.3×10^{-1}	1.4×10^{-1}	1.4×10^{-1}
^{232}Th	2.3×10^{-2}											
^{238}U	2.4×10^{-2}											
^{237}Np	2.7×10^{-2}											
^{239}Pu	3.0×10^{-2}											
^{240}Pu	3.0×10^{-2}											
^{241}Am	3.2×10^{-2}											

[a] Assumes a homogeneous activity distribution in the organism.
[b] Assumes an RBE of 10 for alpha and 1 for beta.

(c) Relative biological effectiveness

122. The effects of radiation exposure on biota depend not only on the absorbed dose, but also on the type or quality of the radiation. For example, alpha particles and neutrons can produce observable damage at much lower absorbed doses than beta or gamma radiation. Thus, the absorbed dose (in gray) is often multiplied by a factor in order to account for the RBE of the quality of the radiation.

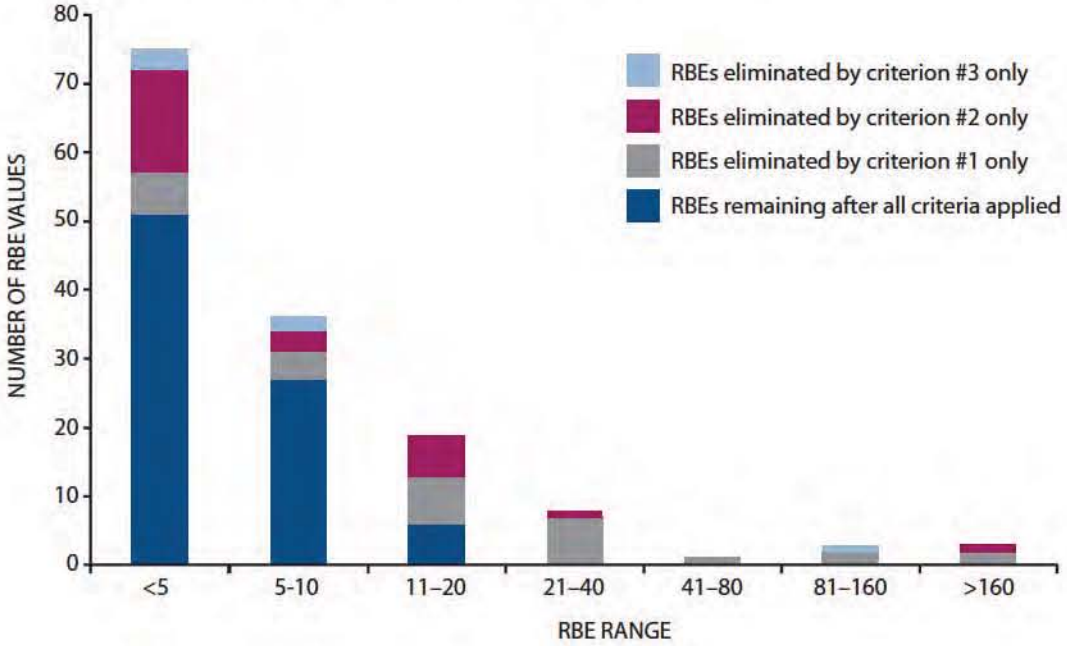
123. A number of authors have evaluated the data on the RBE of different types of radiation [A1, C1, E2, F4, T7, U4, U26]. Nominal values for the factor to reflect the RBE of alpha particles derived from these reviews are summarized in table 16. The experimental values of RBE are specific to the endpoint studied, the biological, environmental and exposure conditions (e.g. reference radiation, dose rate, and dose) amongst other factors. Thus, as noted in a FASSET report [F4], it is difficult to develop a generally valid factor to reflect the RBE for different radiation qualities for use in an environmental risk assessment. The ACRP [A1] and FASSET [F4] have therefore proposed ranges of values for general application. Both selected a factor of 10 to reflect the RBE for alpha particles, the ACRP, citing references [K2, T7, U4], referring to it as a notional central value, and FASSET as a value "to illustrate" the impact of the RBE for an internally deposited alpha emitter.

Table 16. Modifying factors to reflect the RBE of alpha radiation for deterministic effects on non-human biota (relative to low-LET radiation)

Source	*Nominal value*	*Comment*
[N1]	1	Built-in conservatism in dose model
[I4]	20	Numerically the same as the radiation weighting factor used in the protection of humans
[B22]	2–10	Non-stochastic effect of neutrons and heavy ions
[U4]	5	Average for deterministic effects
[T7]	10	Deterministic population-relevant endpoints
[C1]	20	Likely to be conservative for deterministic effects
[E3]	40	Includes studies with high RBE values
[E12]	<35	Based on concentrations in the whole body
[A1]	5–20 (10)	5–10 deterministic effects (cell-killing, reproductive) 10–20 cancer, chromosome abnormalities 10 nominal central value
[F4]	5–50 (10)	10 to illustrate the effect of the alpha RBE
[K19]	<7 to <35	Upper bound of estimate of RBE

124. Chambers et al. [C5] reported a review of the literature on experimentally determined RBEs for internally deposited alpha-emitting radionuclides. The relevance of each experimental result in selecting a factor to reflect the RBE for alpha particles was judged on the basis of pre-established criteria. They recommended a nominal factor of 5 to reflect the RBE for alpha particles for population-relevant deterministic and stochastic endpoints but, to reflect the limitations in the experimental data, they also suggested uncertainty ranges of 1–10 and 1–20 for population-relevant deterministic and stochastic endpoints, respectively. The data developed by Chambers et al. [C5] after application of their evaluation criteria are summarized in figure IX. Three evaluation criteria were used in reference [C5]. Criterion 1 required the dosimetric conditions to be sufficiently well defined and not peculiar to the source of radiation. Criterion 2 required the dose–effect relationships to be sufficiently well known so that the results from the dose rates used experimentally can be applied to effects that may occur with environmental dose rates. Criterion 3 required the experimental uncertainties to be discussed by the authors of the original studies.

Figure IX. Application of the criteria to the distribution of RBEs (all endpoints) [C5]



125. Knowles [K19] reported on experimental studies on groups of zebra fish that were exposed from an early age to different dose rates of gamma and alpha radiation (the latter was from ^{210}Po). Among the gamma-irradiated fish, only those in the highest dose-rate group (7,400 mGy/h) showed radiation-related damage. No groups of alpha-irradiated fish showed evidence of radiation-induced reduction in egg production even though autoradiographs showed concentrations of ^{210}Po in the testes and ovaries. Since the highest alpha dose rate (214 mGy/h) showed no effect, comparison with the gamma dose rate of 7,400 mGy/h, which caused egg production to cease, resulted in only upper limits to the RBE. These were calculated to be in the range of <7 to <20 based on ovary concentrations and <35 based on whole body concentrations. The authors suggested that the RBEs derived from their work provide the best available (upper bound) estimates for a population-relevant effect for fish.

126. The ACRP [A1] considered tritium beta radiation because the low velocity of the beta particles (maximum energy = 18.6 keV) results in a relatively high LET over a short path length. It has an LET very similar to that of 70 keV photons, which are representative of the X-rays used in radiobiological research and in diagnostic medicine [M6]. In their review of the effects of tritiated water (HTO) in mammals and fish, Environment Canada in their Priority Substances List (PSL2) [E3] listed tritium RBE values ranging from 1.7 to 3.8, with gamma rays from ^{60}Co or ^{137}Cs being used as the reference radiation. Based on this, they recommended a factor of 3 to reflect the RBE of beta radiation from tritium. Research conducted at Atomic Energy of Canada Ltd. on breast cancers in female rats [G1] and on myeloid leukaemia in male mice indicated an RBE value of 1.2 for tritium, with X-rays being used as the reference radiation. The difference between these values is largely the result of the choice of reference radiation. Sinclair [S8] has shown that, at low doses, X-rays are about twice as effective as gamma rays in producing damage. Hence, the radiation from tritium has an effectiveness for biological damage in the higher part of the range expected for the gamma and X-ray photon energies likely to be experienced in the environment. Citing Straume and Carsten [S9] amongst others, the ACRP concluded that for the dosimetry of non-human species, where the endpoints are usually deterministic in nature, a reasonable average factor to reflect the RBE of beta particles may be 2 with a range of 1–3, depending on the endpoint being assessed [A1].

127. A number of studies suggested that beta radiation with energies below 10 keV has a higher RBE than electrons with energies above 10 keV [M10, S9]. Straume and Carsten [S9] reviewed 33 studies of the RBE of tritium beta particles and found arithmetic means of 1.8 based on X-rays as the reference radiation, and 2.3 with ^{137}Cs or ^{60}Co gamma rays as the reference radiation. Most of these studies related to deterministic effects. Moiseenko et al. [M10] considered an appropriate factor to reflect the RBE of beta particles from tritium (mean beta energy <10 keV) to be between 2 and 3. The UK Health Protection Agency (HPA) [H21] reviewed the RBE studies on tritium beta particles along with a wide variety of experimental studies using X-rays and gamma rays as reference radiations and noted that the RBEs generally ranged from 1 to 2 when compared to orthovoltage X-rays and from 2 to 3 when compared to gamma rays [H21]. Little and Lambert [L9] also reviewed the experimental studies of cancer induction, chromosomal aberration, cell death and various other endpoints and arrived at similar conclusions for the RBE of tritium in water.

128. In order to illustrate the effect of the radiation quality of emissions from internally deposited radionuclides, the FASSET programme recommended the use of a factor of 10 to reflect the RBE of alpha radiation, 3 for low-energy beta radiation (E < 10 keV), and 1 for both beta radiation with energies greater than 10 keV and for gamma radiation [F4, L4].

129. The Committee, in its UNSCEAR 1996 Report [U4], recommended a nominal factor of 5 to reflect the RBE for internally deposited alpha emitters. The Committee now recommends a nominal (generic) factor of 10 to reflect the RBE for internally deposited alpha radiation. For beta and gamma radiation, the Committee recommends a nominal (generic) factor of 1 to reflect the RBE. However, it should be understood that the most appropriate factor to reflect the RBE for low-energy (<10 keV) beta radiation remains an open question and ought to be the subject of future research. These recommended values to reflect the RBE are intended to apply on a generic basis across all organisms and endpoints. Where appropriate scientific information specific to a particular organism and endpoint exists, such information is preferred.

(d) Dose rates for internal exposure

130. The dose from unit exposure of the selected reference organisms is estimated from the weighted absorbed dose rate due to external exposure arising from deposits in the ground and that due to internal exposure. Weighted absorbed dose rates to the reference organisms normalized for continuous exposure to 1 Bq/m^3 in air for each radionuclide are given in table 17. These weighted absorbed dose rates were calculated assuming the factors to reflect the RBE recommended by the Committee. Table 18 summarizes the ratios of weighted to non-weighted normalized total doses. The results are particularly sensitive to the choice of factor, especially for radiation from the actinides and tritium. The contributions of weighted internal doses to the total normalized doses are close to or above 90%, which indicates that internal exposure is the dominant pathway.

131. These annual doses took account of external exposure as well as internal exposure via inhalation and ingestion. They are compared with the weighted absorbed doses to biota in table 19. The ranges given in the table for biota reflect the variations between the different reference organisms considered. This comparison has however some inherent limitations. The values for humans are expressed in terms of annual effective dose, whereas the values for biota are in terms of weighted absorbed dose and were estimated assuming a homogeneous distribution of the radionuclide in the organism. Furthermore, the annual effective doses per unit deposition to humans were based on a radiation weighting factor of 20 for alpha particles, whereas the weighted absorbed doses to biota were based on a factor of 10 to reflect the RBE for alpha particles. Further still, the values for humans reflect largely the transfer of radionuclides through agricultural ecosystems, whereas the values for biota are more typical of the transfer in forests and semi-natural ecosystems.

132. Despite these differences, the estimated normalized effective doses to humans and the weighted absorbed doses to biota are about the same order of magnitude, except in the cases of ^{129}I and ^{131}I. These exceptions are probably due to the special importance of radiation exposure of the human thyroid in evaluating effective dose, which has no counterpart in the dosimetry for biota. Thus, apart from these exceptions, the comparison indicates that for similar levels of radionuclides in the environment, the effective doses to humans and the weighted absorbed doses to biota are comparable.

Table 17. Normalized weighted absorbed dose rates per unit activity concentration to various biota from internal exposure

Radionuclide	*Weighted dose rate per unit activity concentration (μGy/h per Bq/m³)*							
	Earthworm	*Rat*	*Deer*	*Duck*	*Frog*	*Bee*	*Grass*	*Pine tree*
^{3}H	1.7×10^{-19}	1.7×10^{-19}	1.7×10^{-19}	1.7×10^{-19}	1.7×10^{-19}	1.7×10^{-19}	1.7×10^{-19}	1.7×10^{-19}
^{36}Cl	5.6×10^{-15}	3.3×10^{-13}	3.3×10^{-13}	3.3×10^{-13}	3.3×10^{-13}	9.3×10^{-15}	5.4×10^{-13}	5.5×10^{-14}
^{90}Sr	6.1×10^{-15}	1.2×10^{-12}	1.3×10^{-12}	7.3×10^{-13}	7.5×10^{-13}	5.1×10^{-14}	1.8×10^{-13}	4.3×10^{-13}
^{99}Tc	4.4×10^{-15}	4.4×10^{-15}	4.4×10^{-15}	4.4×10^{-15}	4.4×10^{-15}	4.4×10^{-15}	2.4×10^{-13}	4.8×10^{-15}
^{129}I	2.3×10^{-14}	4.8×10^{-14}	4.9×10^{-14}	4.5×10^{-14}	4.0×10^{-14}	2.9×10^{-14}	1.7×10^{-14}	2.1×10^{-14}
^{131}I	4.6×10^{-16}	1.3×10^{-15}	1.8×10^{-15}	4.2×10^{-16}	3.4×10^{-16}	1.9×10^{-16}	1.6×10^{-15}	2.8×10^{-15}
^{134}Cs	1.7×10^{-13}	2.7×10^{-13}	4.5×10^{-13}	9.7×10^{-14}	8.1×10^{-14}	6.6×10^{-14}	8.7×10^{-14}	1.2×10^{-13}
^{135}Cs	8.3×10^{-15}	2.7×10^{-13}	2.7×10^{-13}	7.1×10^{-14}	5.4×10^{-14}	5.0×10^{-15}	6.7×10^{-14}	1.7×10^{-14}
^{137}Cs	4.7×10^{-13}	4.4×10^{-13}	1.1×10^{-13}	1.9×10^{-13}	2.0×10^{-13}	2.0×10^{-13}	1.9×10^{-13}	2.3×10^{-13}
^{210}Pb	2.9×10^{-14}	9.4×10^{-15}	9.2×10^{-15}	1.2×10^{-14}	1.2×10^{-14}	1.1×10^{-14}	1.3×10^{-14}	8.7×10^{-16}
^{226}Ra	5.0×10^{-12}	2.1×10^{-12}	4.7×10^{-13}	8.9×10^{-13}	9.3×10^{-13}	3.7×10^{-12}	1.5×10^{-11}	1.3×10^{-12}
^{232}Th	4.9×10^{-13}	6.4×10^{-15}	5.6×10^{-15}	2.8×10^{-14}	2.8×10^{-14}	4.9×10^{-13}	8.4×10^{-13}	4.0×10^{-13}
^{238}U	5.8×10^{-12}	2.4×10^{-12}	2.4×10^{-12}	2.4×10^{-12}	2.4×10^{-12}	7.5×10^{-12}	9.9×10^{-13}	1.8×10^{-11}
^{237}Np	1.9×10^{-12}	1.5×10^{-12}	1.5×10^{-12}	1.5×10^{-12}	1.5×10^{-12}	4.0×10^{-12}	4.1×10^{-12}	4.9×10^{-12}
^{239}Pu	2.1×10^{-12}	1.7×10^{-12}	1.7×10^{-12}	1.7×10^{-12}	1.7×10^{-12}	4.4×10^{-12}	4.5×10^{-12}	5.4×10^{-12}
^{240}Pu	7.2×10^{-12}	2.9×10^{-12}	2.9×10^{-12}	2.9×10^{-12}	2.9×10^{-12}	9.3×10^{-12}	3.9×10^{-12}	3.1×10^{-12}
^{241}Am	8.8×10^{-14}	8.6×10^{-14}	7.6×10^{-14}	7.9×10^{-14}	8.0×10^{-14}	8.0×10^{-14}	3.7×10^{-12}	3.3×10^{-12}

Table 18. Ratio of weighted and unweighted doses

Radionuclide	Ratio of weighted dose/unweighted dose[a]							
	Earthworm	Rat	Deer	Duck	Frog	Bee	Grass	Pine tree
^{3}H	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0
^{36}Cl	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0
^{90}Sr	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0
^{99}Tc	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0
^{129}I	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0
^{131}I	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0
^{134}Cs	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0
^{135}Cs	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0
^{137}Cs	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0
^{210}Pb	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0
^{226}Ra	2.0	1.0	1.1	1.2	1.1	3.3	7	2
^{232}Th	9.9	7.1	9.5	9.7	9.7	10	10	10
^{238}U	10	10	10	10	10	10	10	10
^{237}Np	9.1	8.9	9.6	9.4	9.3	9.6	9.6	9.7
^{239}Pu	10	10	10	10	10	10	10	10
^{240}Pu	10	10	10	10	10	10	10	10
^{241}Am	4.1	4.3	7.9	6.1	5.9	5.9	10	10

a Factors to reflect the RBE: alpha radiation, 10; beta and gamma radiation, 1.

Table 19. Comparison of doses to biota and humans, normalized for unit deposition to terrestrial ecosystems

Radionuclide	Biota (range)[a] Normalized weighted absorbed dose rate ($Gy\ a^{-1}$ per $Bq\ m^{-2}\ a^{-1}$)		Humans[b] Normalized effective dose rate ($Sv\ a^{-1}$ per $Bq\ m^{-2}\ a^{-1}$)
	Minimum	Maximum	
^{90}Sr	6.2×10^{-9}	1.3×10^{-6}	4.7×10^{-7}
^{99}Tc	4.4×10^{-9}	2.5×10^{-7}	1.8×10^{-8}
^{129}I	1.7×10^{-8}	5.0×10^{-8}	6.3×10^{-7}
^{131}I	2.0×10^{-10}	2.8×10^{-9}	1.0×10^{-7}
^{134}Cs	6.7×10^{-8}	4.6×10^{-7}	1.3×10^{-7}
^{135}Cs	5.1×10^{-9}	2.8×10^{-7}	1.2×10^{-8}
^{137}Cs	1.1×10^{-7}	4.8×10^{-7}	1.3×10^{-7}
^{210}Pb	8.9×10^{-10}	2.9×10^{-8}	2.5×10^{-6}
^{226}Ra	4.8×10^{-7}	1.5×10^{-5}	1.6×10^{-6}
^{232}Th	5.6×10^{-9}	8.5×10^{-7}	1.2×10^{-6}
^{238}U	1.0×10^{-6}	1.8×10^{-5}	6.0×10^{-7}
^{237}Np	1.5×10^{-6}	5.0×10^{-6}	4.9×10^{-7}
^{239}Pu	1.7×10^{-6}	5.5×10^{-6}	6.8×10^{-7}
^{240}Pu	3.0×10^{-6}	9.5×10^{-6}	6.8×10^{-7}
^{241}Am	7.7×10^{-8}	3.8×10^{-6}	5.8×10^{-7}

a Range represents the minimum and maximum among the organisms considered.
b Calculated according to [I11].

3. Doses to non-human biota

(a) Calculation of doses to biota

133. In terrestrial environments, the most important source of radiation exposure as a consequence of discharges of radionuclides to the environment is due to deposition on soil. Radionuclides present in soil are generally a persistent radiation source for all terrestrial biota. Aquatic organisms are irradiated externally by the activity in water and, in the cases of bottom dwellers and benthic organisms, the activity in sediments, and internally by incorporated radionuclides. The dose rate, D, can be calculated according to:

$$D = \sum_r \left[DCC_{ext,r} \cdot C_{soil,water,r} + DCC_{int,r} \cdot C_{biota,r} \right] \quad (5)$$

where $DCC_{ext,r}$ is the DCC for external exposure to radionuclide r (μGy/h per Bq/kg); $C_{soil,water,r}$ is the activity concentration of radionuclide r in soil or water (Bq/kg); $DCC_{int,r}$ is the DCC for internal exposure to radionuclide r (μGy/h per Bq/kg); and $C_{biota,r}$ is the internal activity concentration of radionuclide r in biota (flora or fauna) (Bq/kg).

(b) Activities in environmental media

134. In the absence of measurements, in order to evaluate equation (5), the activity concentrations, $C_{soil,water,r}$, and $C_{biota,r}$, have to be estimated. Assuming a constant discharge of radionuclides over a period of 50 years, the activity in soil for the last year of that period is calculated as indicated in reference [I11]:

$$C_{s,r} = \frac{D_{tot,r}}{(\lambda_r + \lambda_m) \cdot m_s} \cdot [1 - \exp(-(\lambda_r + \lambda_m) \cdot t_e)] \quad (6)$$

where $C_{s,r}$ is the activity concentration in soil (Bq/kg); $D_{tot,r}$ is the total (wet plus dry) deposition rate to soil (Bq m^{-2} a^{-1}); m_s is the mass of the upper soil layer (kg/m^2); λ_r is the radioactive decay constant (a^{-1}); λ_m is the loss rate from the upper soil layer (a^{-1}); and t_e is the discharge period (50 a).

135. The total deposition is calculated as the sum of dry ($D_{dry,r}$) and wet deposition ($D_{wet,r}$). The activity concentration in flora, $C_{flora,r}$, is estimated by taking into account direct deposition on the foliage and uptake from soil according to reference [I11]:

$$C_{flora,r} = \frac{D_{dry,r} + f_w \cdot D_{wet,r}}{(\lambda_{w,r} + \lambda_r) \cdot b} \cdot [1 - \exp(-(\lambda_{w,r} + \lambda_r) \cdot t_w)] + C_{s,r} \cdot TF_{flora,r} \quad (7)$$

where f_w is the interception fraction (dimensionless); b is the standing biomass (kg/m^2); λ_w is the activity loss rate from plants due to weathering (a^{-1}); t_w is the exposure time (a); and $TF_{flora,r}$ is the transfer factor from soil to flora (Bq/kg flora per Bq/kg soil).

136. The activity concentration in reference fauna is estimated from the soil concentration and the soil–fauna transfer factor as follows:

$$C_{fauna,r} = C_{s,r} \cdot TF_{fauna,r} \quad (8)$$

where $TF_{fauna,r}$ is the soil–fauna transfer factor (Bq/kg fauna per Bq/kg soil).

137. The habitats of the reference fauna are differentiated according to whether the organisms live in or above soil. DCCs for species living in soil are expressed in units of μGy/h per Bq/kg and are based on the assumption that the organism lives in the centre of a slab containing radionuclides uniformly distributed to a depth of 50 cm. For organisms living on soil, it is assumed that radionuclides are homogeneously distributed to a depth of 10 cm; the DCCs in this case have units of μGy/h per Bq/m^2.

138. The estimation of the activity concentration of a radionuclide in aquatic biota ($C_{aqua\ biota}$) is usually obtained from the activity concentration in water (C_{water}) and the concentration factor ($CF_{water-biota}$) according to:

$$C_{aqua\ biota} = C_{water,r} \cdot CF_{water-biota,r} \quad (9)$$

139. As outlined above, the exposure due to incorporated radionuclides is determined by the size and geometry of the organism, the radionuclide distribution, and the type and energy of the emitted radiation. Currently, DCCs are not available for specific target organs in the reference organisms; the DCCs for internal exposure are therefore based on the assumption that the radionuclides are homogeneously distributed throughout the organism [T10].

(c) Doses to marine organisms and to humans due to consumption of marine food

140. As an example of the calculations of exposures of aquatic organisms, the exposures to marine organisms are calculated assuming a radionuclide concentration in water of 1 Bq/m^3 and applying the appropriate concentration factor for water–biota in table 13 and the appropriate DCCs given in tables 14 and 15. The weighted absorbed dose rates to flatfish, crab and brown seaweed are summarized in table 20. For all radionuclides considered, the dose rates to biota are almost completely a result of internal exposure. For comparison, the effective dose rates to an adult human are given assuming an annual fish intake of 20 kg. In general, the effective dose rates to humans are much less than the weighted absorbed dose rates to biota for a unit activity concentration of a radionuclide in marine water.

Table 20. Comparison of doses to non-human biota and humans, normalized to an activity concentration in marine water of 1 Bq/m³

Radionuclide	*Non-human biota*			*Humans*[a]
	Weighted absorbed dose rate (μGy/h per Bq/m³)			*Normalized effective dose rate (μSv/h per Bq/m³)*
	Flatfish	*Crab*	*Macroalgae*	
^{3}H	3.3×10^{-9}	3.3×10^{-9}	3.3×10^{-9}	4.1×10^{-11}
^{36}Cl	9.6×10^{-9}	9.6×10^{-9}	7.0×10^{-9}	1.3×10^{-10}
^{90}Sr	1.8×10^{-6}	6.3×10^{-6}	4.5×10^{-6}	1.9×10^{-7}
^{99}Tc	4.6×10^{-6}	5.8×10^{-5}	1.7×10^{-4}	1.2×10^{-7}
^{129}I	3.8×10^{-7}	4.3×10^{-6}	3.8×10^{-4}	2.3×10^{-6}
^{131}I	1.2×10^{-6}	1.4×10^{-5}	1.0×10^{-3}	4.5×10^{-7}
^{134}Cs	1.9×10^{-5}	6.9×10^{-6}	4.9×10^{-6}	4.3×10^{-6}
^{135}Cs	3.9×10^{-6}	1.2×10^{-6}	1.9×10^{-5}	4.6×10^{-7}
^{137}Cs	1.8×10^{-5}	5.7×10^{-6}	6.5×10^{-6}	3.0×10^{-6}
^{210}Pb	4.8×10^{-5}	2.3×10^{-3}	8.0×10^{-4}	3.2×10^{-4}
^{226}Ra	1.3×10^{-2}	1.4×10^{-2}	1.4×10^{-2}	6.4×10^{-5}
^{232}Th	1.4×10^{-2}	2.3×10^{-2}	4.6×10^{-3}	3.2×10^{-4}
^{238}U	2.4×10^{-5}	2.4×10^{-4}	2.4×10^{-3}	1.0×10^{-7}
^{237}Np	2.7×10^{-3}	2.7×10^{-3}	1.4×10^{-3}	2.5×10^{-5}
^{239}Pu	3.0×10^{-3}	6.0×10^{-3}	1.2×10^{-1}	5.7×10^{-5}
^{240}Pu	3.0×10^{-3}	6.0×10^{-3}	1.2×10^{-1}	5.7×10^{-5}
^{241}Am	3.2×10^{-3}	1.3×10^{-2}	2.5×10^{-1}	4.6×10^{-5}

[a] For an intake of marine fish of 20 kg/a.

4. Conclusions

141. In this section, approaches have been described for the assessment of exposures of flora and fauna to radiation from natural background levels of radionuclides or regulated discharges of radionuclides to the environment. The models cover two major fields. One is concerned with the transport processes of radionuclides from the source to plants and animals, to which approaches may be applied that are similar to those used to assess the exposures of humans. In the terrestrial environment, these are mainly atmospheric dispersion, deposition, interception, weathering and uptake from soil. For discharges to aquatic systems, models can be used that describe dispersion, dilution, sedimentation and uptake by freshwater or marine organisms.

142. There are major differences in the dosimetry involved in the assessment of the exposures of humans and non-human biota. The current approaches for biota rely on the mean activity concentrations in the whole organism rather than on those in distinct organs or tissues. Thus, the calculated absorbed doses are to the whole organism. There is an ongoing discussion about the appropriate factors to be applied in order to account for the different RBEs of the different kinds of radiation involved. Example calculations in this annex show that the estimated weighted absorbed doses from exposure to alpha radiation are sensitive to the value of the factor used. This is relevant to the assessment of doses to biota both as a result of radioactive discharges from a nuclear site and as a result of exposure to radiation from radionuclides that are naturally present in the environment.

143. The estimated doses to biota are compared in this annex with those to humans in accordance with the approach given in reference [U3]. The comparison shows that the weighted absorbed doses to terrestrial non-human biota and the effective doses to humans are generally of a similar order of magnitude, for a given level of environmental contamination by radionuclides. The weighted absorbed doses to marine biota are, in general, considerably higher than the effective doses to humans (for whom an annual consumption of marine fish of 20 kg is assumed for illustrative purposes).

144. The results of the dosimetric calculations presented in this annex are based on stylized models of ecosystems using average values for most of the model parameters. Thus, they do not accurately reflect the variability of ecosystems and the processes present in nature that control the environmental mobility of radionuclides. In addition, the exposures due to the various sources of natural background radiation and their variabilities would have to be included if the results presented in this annex were to be used in a site-specific assessment. As indicated earlier, there are substantial uncertainties associated with the estimation of dose rates to non-human biota, including those associated with the environmental pathways (such as in the values of the transfer factors) and those related to dosimetric issues.

II. SUMMARY OF DOSE–EFFECTS DATA FROM THE UNSCEAR 1996 REPORT

145. In the absence of reports of obvious deleterious effects on other organisms from exposure to environmental radiation, whether of natural origin or due to the controlled discharges of radionuclides to the environment, it had generally been accepted that priority should be given to evaluating the potential consequences for humans (which are among the most radiosensitive mammalian species) and to providing a sound basis for protecting human health. By 1996, this position had, however, been questioned [D1, T1], and at least one situation (namely deep-sea sediments, an environment very remote from man) had been identified where the above accepted priority could be incorrect [I3]. In response to such concerns, the Committee noted that the impact of radiation exposure of non-human biota had been studied in a number of situations [I2, I3, I4, M1, N1, W1, W2] and considered that it was appropriate to conduct an independent review of the matter and to summarize the state of knowledge existing at that time. The UNSCEAR 1996 Report [U4] took account of the earlier reviews and studies and the Committee's summaries of the radiobiological work carried out over the previous 50 years.

146. In its 1996 report, the Committee noted that there was a fundamental difference in the approaches to the protection of humans and non-human biota from the effects of exposure to ionizing radiation. For humans, ethical considerations had made the individual the principal object of protection. This meant, in practice, that any incremental risk to the individual arising from increased radiation exposure was to be constrained below some level that society judged to be acceptable; this level of risk, although small, was not zero [I5]. For non-human biota, the populations of the biota were considered to be important and protection from a significantly increased risk to each population arising from radiation exposure might be the appropriate objective. Exceptions might be populations of small size (rare species) or those reproducing slowly (i.e. with long generation times and/or low fecundity) for which protective measures might be more appropriately targeted at the level of the individual organism. The Committee noted that there could not be any effect at the population level (or at the higher levels of community and ecosystem) if there were no effects on the individual organisms constituting the different populations. It went on to suggest that radiation-induced effects on some members of a population would not necessarily have any significant consequences for the population as a whole.

147. The Committee noted that natural populations of organisms existed in a state of dynamic equilibrium within their communities and environments and that exposure to ionizing radiation was but one of the stresses that may affect this equilibrium. The incremental radiation exposure from human activities could not, therefore, be considered in isolation from other sources of stress, whether natural (e.g. climate, altitude, or volcanic activity) or of human origin (e.g. synthetic chemical toxins, oil discharges, exploitation for food or sport, or habitat destruction). When (as is not uncommon) ionizing radiation and chemicals, both resulting from human activities, acted together on a population, the difficult problem arose of correctly attributing any observed response to a specific cause.

148. The objective of the UNSCEAR 1996 Report on the "Effects of radiation on the environment" [U4] was to summarize and review information on:

- The exposures (actual or potential) of organisms in their natural habitats to the natural background radiation, to radionuclides discharged into the environment in a controlled manner from industrial activities, and to radionuclides released as a consequence of accidents; and
- The responses of plants and animals, both as individuals and as populations, to acute and chronic irradiation.

149. The Committee hoped that its review would assist national and international bodies to select appropriate criteria for the radiological protection of natural populations, communities and ecosystems. The following paragraphs recapitulate the information available to the Committee in 1996.

A. Dosimetry for environmental exposures

150. As discussed in the annex to the UNSCEAR 1996 Report [U4], reliable determination of the dose rate to organisms is essential for assessing the potential or actual impacts of contaminant radionuclides in the environment. The Committee noted that "this simple statement conceals a multitude of difficulties that prevent the easy achievement of that estimation". In practice, it is necessary to make simplifying assumptions, with the degree of simplification depending on the purpose of the assessment. For example, for the purpose of screening, the concept of a single generic biota that represented all plants and animals had been used [A2]. More sophisticated models attempted to account for the dose distributions within reference organisms of assumed shapes and sizes and the fraction of radiation being absorbed within the organism [W2]. The Committee's views on dosimetry for estimating the exposure of biota based on what was known in the UNSCEAR 1996 Report [U4] are summarized below.

151. A dosimetric model is essentially a mathematical construction that allows the energy deposition in a defined target to be estimated from a given radionuclide (source) distribution. The model was often derived using theoretical or empirical functions that described the distribution of dose about a point source [B2, B3, L1, W2]. The dose at a point in the target was then obtained by integrating the point source dose distribution function over the defined radionuclide source, either internal or external to the organism. This

procedure was frequently simplified by using ideal geometries (spheres, ellipsoids, etc.) of appropriate size to represent the target and by assuming that the radionuclide distribution was uniform (over a surface or through a volume) or varied in a way that could be described by a simple mathematical expression (e.g. an exponential decline in radionuclide concentration with depth in soil or sediment). Alternatively, Monte-Carlo calculations had been used to determine the absorbed fractions of energy for a variety of source and target geometries [B4, E2]. These data could be used, either directly or with interpolation (or, to a lesser extent, with extrapolation) for geometries that could represent targets of environmental concern. In principle, these procedures could be adapted for use in estimating doses to terrestrial and aquatic organisms, from both the plant and animal kingdoms, for both internal and external sources of radiation.

152. Dosimetric models had been developed to take account of the radiation type; the specific geometry of the target (e.g. the whole body, the gonads, the developing embryo or the plant meristem); and the source of exposure (e.g. radionuclides accumulated in body tissues, adsorbed onto the body surface or distributed in the underlying soil). Clearly, it was not possible to consider all organisms, and there were limitations in the basic data that were available as input to the models (e.g. the spatial and temporal distributions of radionuclides both within the organism and in the external environment). Additional sources of complexity arose from the behaviour of mobile organisms, particularly some aquatic organisms and many insects, which inhabit different environmental niches at different stages of their life cycles. Thus, the models had to be simplified and generalized without undue loss of the realism that is essential for a valid estimation of dose.

153. The presence of an alpha particle component in the total absorbed dose rate to a tissue in a plant or animal raised the question of how to take account of the probably greater effectiveness of this type (quality) of radiation in producing biological damage. The RBEs of different qualities of radiation had been very critically examined for the purposes of human radiation protection. Each component of the absorbed dose to a tissue or organ was weighted by a factor which took account of the RBE of the radiation involved [I5]. It seemed reasonable to apply a similar approach to the radiation dosimetry for organisms other than man. In practice, however, there were circumstances that altered the detailed application of this approach. In the human case, the major concern had been with the induction of stochastic effects (principally cancer) at low doses and dose rates. For alpha radiation, experimental determinations of the RBE had led to a recommended radiation weighting factor of 20 for the purpose of human radiation protection. In the case of wild animals, however, the Committee assumed that it was likely that deterministic effects were of greater significance. For alpha radiation, the experimental data for animals indicated that a lower factor to reflect the RBE would be more appropriate; the factor to reflect the RBE of beta and gamma radiations would however be numerically the same as the radiation weighting factor used in human radiation protection. On the assumption that mammals are the most sensitive species, these values could be applied to other taxonomic groups.

154. In its 1996 UNSCEAR Report [U4], the Committee assumed that these factors would also apply to effects on plants, although there were no definitive experimental data to support this. In the absence of protection quantities (equivalent and effective dose) for non-human organisms, the absorbed doses from low-LET radiation (beta particles, X-rays and gamma rays) and from high-LET radiation (alpha particles) were assessed and specified separately in the UNSCEAR 1996 Report [U4]. The absorbed doses retained the unit, joule per kilogram (J/kg), with the special name gray (Gy).

155. An IAEA technical report [I4] provided estimates of the dose rates to terrestrial plants due to radionuclides deposited following discharges to the atmosphere. The model, PATHWAY [W3], developed to estimate doses to humans, had been used to derive the equilibrium concentrations of radionuclides in plants and animals for the limiting case in which humans, while living on the land, breathing the air over it and eating the food produced from it, would receive an annual effective dose of 1 mSv. To estimate the dose to plants from internal sources, it was assumed that the energy of alpha and beta particles would be totally absorbed (except for emissions from ^{32}P, which would be 50% absorbed) and that 10% of the gamma-ray energy would be absorbed. An additional degree of conservatism was provided by using estimates of the radionuclide concentrations in plant tissue on a dry weight basis (which are 5–10 times higher than on a wet weight basis) to calculate the absorbed dose rates to living (i.e. "wet") plant tissue. The results are given in table 21. As these estimates had been made using a radioecological model and a scenario designed for calculating exposures to humans, the calculated exposures of non-human species should be interpreted cautiously.

156. The annex of the UNSCEAR 1996 Report [U4] noted that there have been fewer estimates of the potential exposures of fully terrestrial animals than of animals occupying semi or fully aquatic niches. This was thought to be a reflection of the greater use that had been made of aquatic systems for the discharge of radioactive waste.

157. The annex of the UNSCEAR 1996 Report [U4] suggested that naturally occurring alpha-emitting radionuclides appeared to be the most significant sources of background radiation exposure for the majority of wild organisms.

158. In its 1996 report, the Committee considered that the data on the radiation exposures of non-human biota due to both natural background radiation and contaminant radionuclides were incomplete, more in some areas than in others. The Committee also noted that the aquatic environment was probably the most thoroughly studied environment up to that time [I2, I3, I7, N1, N2, W1], even with the substantial

generalizations that had had to be made, particularly with respect to the range of organisms that could reasonably be considered [I3]. As had been emphasized elsewhere [I3, I6], the limiting factor was not the development of an appropriate dosimetric model for a particular organism but rather the acquisition of essential input data on the temporal and spatial distributions of the radionuclides both external to and within the organism. Although dynamic models had been employed to describe the dispersion and dilution of radionuclides in a water body, related phenomena (e.g. transfers to sediments and biological tissues) were almost always modelled as equilibrium processes, i.e. using simple distribution coefficients and (whole-body) concentration factors. This simplification largely neglected the temporal variations in dose rate due, for example, to short-term fluctuations in discharge rate, differing stages in the life cycle, and behavioural and short-term environmental processes (e.g. seasonality). As a consequence, while the estimated absorbed dose rate might be a reasonable indication of the general magnitude of the actual environmental value, the Committee considered that it did not provide a very secure basis for evaluating total doses over time.

Table 21. Estimated dose rates to organisms from controlled discharges of radionuclides that would each result in an annual dose of 1 mSv to humans residing in the same environment
Table 6 of UNSCEAR 1996 Report [U4]; based on [I4, N1]

Radionuclide	*Dose rate (μGy/h)*		
	Plants[a]	*Animals*[a,b]	*Fish*[c]
^{3}H	5.8	5.8	0.59
^{14}C	18	11	
^{32}P	32	28	4.8
^{60}Co			0.53
^{90}Sr	2.0	0.042	67
^{95}Zr	38	2.0	
^{99}Tc			3.8
^{131}I	1.2	0.058	
^{137}Cs	5.4	3.1	0.72
^{226}Ra[d]			3.6
^{235}U[d]			2.6
^{238}U[d]			4.7
^{239}Pu[d]	0.023	0.000 55	0.49
^{241}Am[d]			0.71

[a] Discharges to atmosphere.
[b] Domestic sheep.
[c] Discharges to water (lakes).
[d] High-LET radiation.

159. The Committee also noted that accident situations were by nature quite different from routine situations, particularly in their potential to produce high dose rates and doses to the environment. It concluded that generalization is difficult because the actual exposure regime depends on the types and quantities of radionuclides released, their initial dispersal and deposition patterns, and their redistribution over time in the environment. Following the accident at the Chernobyl nuclear power plant, large quantities of short-lived radionuclides were released, leading to high dose rates in the local area. Total doses up to 100 Gy were delivered to trees (and, by inference, to most other organisms in the locality) over a period of a few days [K1]. This radiation regime might have been characterized as "acute" in that the doses were delivered in periods that were shorter than or comparable to the time taken for severe damage to become apparent. During this initial (acute) phase, the dose rates declined rapidly as the very short-lived radionuclides decayed. The release following the accident in 1957 in the south-eastern Urals was dominated by ^{144}Ce–^{144}Pr (approximately 66%; $t_{½}$ = 285 d) and ^{95}Zr–^{95}Nb (approximately 25%; $t_{½}$ = 65 d). In that case, the dose rates locally were also relatively high during the initial phase (more than 4 mGy/h) but declined more slowly, such that high total doses (causing severe effects, including mortality) could still be accumulated from essentially chronic exposure. Close to the release point, total doses up to 2,000 Gy were experienced [T4]. In the longer term, the exposure regime for the Chernobyl release was dominated by ^{137}Cs ($t_{½}$ = 30 a) and ^{90}Sr ($t_{½}$ = 28.6 a), and that for the south-eastern Urals accident area by ^{90}Sr. In both cases, the exposures were chronic and moderately high, with responses other than mortality becoming significant.

160. Radioactive waste discharges to atmosphere, landfills or aquatic systems from man-made practices entail increased radiation exposure of wild organisms. The incremental radiation exposures are chronic (i.e. continuing) at absorbed dose rates of generally no more than 100 μGy/h, but, very exceptionally, they may reach several thousand microgray per hour. The Committee [U4] noted that these additional

radiation exposures may be greater than the normal range of natural background exposures but generally are within the extreme range of background exposures, if the exceptional cases of areas of uranium and thorium mineralization are included. Given that radioactive waste discharged to the environment will normally be dispersed and diluted, dose rates higher than those due to normal natural background exposure are likely to apply to only a small proportion of the individual organisms in any population and the average dose rate to the population would probably be much lower [W8, W9].

B. Effects of radiation exposure on plants and animals

161. Studies of the effects of ionizing radiation exposure on plants and animals were started immediately following the discovery of X-rays and radioactivity (see, for example, reference [A4]). Since 1945, when the first nuclear detonations were conducted, there was widespread concern about the impact of environmental radiation exposures and interest in the environmental behaviour of radioactive materials. As a result, studies using a wide variety of plant and animal species were performed [A4, B5, C3, P1].

162. The Committee, in its 1996 report [U4], noted that the responses of organisms to radiation exposure were varied and may become manifest at all levels of organization, from individual biomolecules to ecosystems. The significance of a given response depended on the criterion of damage adopted, and it was not to be concluded that a response at one level of organization would necessarily produce a consequential, detectable response at a higher level of organization.

163. The Committee also noted that a population might be defined as all members of a population species [U4]. Alternatively, a population might be considered as an aggregate of inter-breeding individuals of a species occupying a specific location in space and time [S5]. The latter definition is perhaps more useful given the Committee's observation that radiation fields, such as those arising from radioactive waste discharges, generally show large spatial variability, not least because of the often discrete nature of the source, and therefore many members of a population might not receive any significant exposure from a particular source. The natural distributions of most species are inhomogeneous because of the variations in physical, chemical and biological conditions under which the individuals of the species are able to survive, i.e. species are geographically restricted. Thus, it is probable that a more limited, and relevant, definition of a population could be developed for the purposes of environmental impact assessment.

164. The following definition (developed for use in population ecology) has been suggested as a useful basis for discussion and progress [I4]: "A population is a biological unit for study, with a number of varying statistics (e.g. number, density, birth rate, death rate, sex ratio, age distribution), and which derives a biological meaning from the fact that some direct or indirect interactions among its members are more important than those between its members and members of other populations" [B6]. Notwithstanding this definition, it has to be understood that a population of a particular species is always linked to its environment. Such a population would (or could) be a self-sustaining unit, independent of other, geographically separate populations of the same species. However, protection of this population would require that increased radiation exposure did not significantly affect the attributes mentioned in the definition on which the population depended for its maintenance within the normal dynamic range of variation dictated by the interactions of natural physical, chemical and biological factors.

165. These attributes, which could be defined only for populations of organisms and might be taken to be indicators of their health, are nevertheless amalgamations of properties that relate to individuals (in no sense was this meant to imply simple addition). The Committee concluded, in effect, that for a response to radiation exposure at the population level (or, indeed, at any higher level of organization) some clearly detectable effect in individual organisms (i.e. at lower levels of organization) would be expected. This clearly implied that the protection of the population (as the ultimate objective) might be achieved by restricting the exposure of individual organisms to the extent that there are no significant radiation effects on those processes necessary for the maintenance of the population. It is therefore necessary to consider the available information on the effects of radiation exposure (mainly at chronic low dose rates) on the relevant processes in individual organisms, to consider how these responses might translate to an impact on the population, and to examine the results of studies of population responses to deliberate experimental irradiation or to exposure in the environment due to controlled or accidental releases of radionuclides.

166. Examination of the population attributes indicated that the individual responses to radiation exposure likely to be significant at the population level are mortality (affecting age distribution, death rate and density), fertility (birth rate), fecundity (birth rate, age distribution, number and density) and the induction of mutations (birth rate and death rate). These individual responses can be traced to events at the cellular level in specific tissues or organs. An extended summary discussing the processes involved was provided in annex J, "Non-stochastic effects of irradiation", of the UNSCEAR 1982 Report [U9]. There was a substantial body of evidence indicating that the most radiosensitive sites are associated with the cell nucleus, specifically the chromosomes, and that, to a lesser extent, damage to intracellular membranes is additionally involved. The end result is that the cells lose their reproductive potential. For most cell types, at moderate doses, death occurs when the cell attempts to divide; death does not, however, always occur at the first post-exposure division: at doses of a few gray, several division cycles might be successfully completed before death eventually occurs. It was also well known that radiosensitivity varies within the cell cycle, with the greatest sensitivities being apparent at mitosis and the commencement

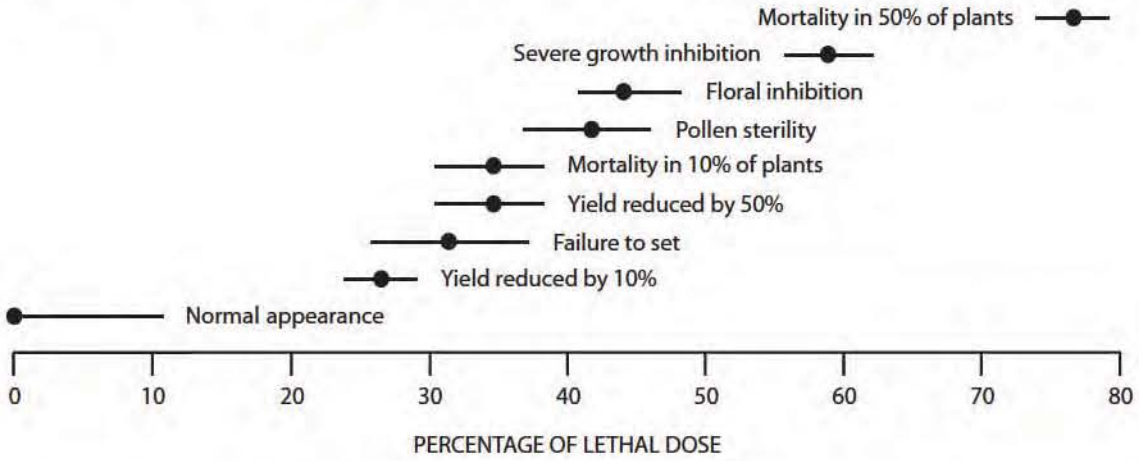
of DNA synthesis [U9]. It followed that the greatest radiosensitivity is likely to be found in cell systems undergoing rapid cell division for either renewal (e.g. spermatogonia) or growth (e.g. plant meristems and the developing embryo); these examples clearly underlie the processes in individual organisms that are important for the maintenance of the population.

167. Fractionation or protraction of exposure to low-LET radiation increases the total dose required to produce a given degree of damage since at low dose rates, the factors responsible for mitigating the response come into play. These include the repair of sublethal damage, the repair of potentially lethal damage, the replacement of killed cells through proliferation of survivors, and other slow repair processes not related to cell repopulation [U9]. Although it was clear to the Committee that repair, in the general sense, is possible, the existence and extent of residual injury was less clear. While such an outcome might be demonstrated for moderate, acute doses, it was not possible to extrapolate these results in order to predict the likely response to low-level exposures extending over a significant fraction of the lifetime of an organism. However, given that genetic mutations might be passed from generation to generation, it was reasonable to suppose that somatic mutations individually consistent with cell survival could occur and accumulate over time until the combined impact might reduce cell viability.

1. Terrestrial plants

168. Radiation injury in plants expresses itself as abnormal shape or appearance, reduced growth or yield, loss of reproductive capacity, wilting and (at high exposures) death [S1]. Acute lethal doses to higher plants ranged from 10 to about 1,000 Gy (approximate mean absorbed doses averaged over the whole plant). The Committee concluded that plants such as mosses, lichens and unicellular species are at one extreme of radiosensitivity being highly resistant to radiation exposure; woody species are at the other extreme being the most sensitive. In 12 species of woody plants assessed 10–14 months after exposure, the lethal doses were found to be in the range of 8–96 Gy [S2]. The pine tree was the most sensitive, experiencing mortality following short-term absorbed doses of about 10 Gy [W5]; growth was severely inhibited at 50–60% of the lethal dose. Floral inhibition was observed at 40–50% of the lethal dose, and failure to set seed at 25–35%. Thus, the capacity of the plant population to maintain itself could be damaged at acute doses lower than those required to cause mortality. Below 10% of the lethal dose, effects were not so apparent and the plants maintained a normal appearance. These general observations for several herbaceous plant species are illustrated in figure X [S3]. Another general relationship was that the dose that reduced survival by 10% (LD_{10}) was roughly equivalent to the dose that reduced the yield by 50% (YD_{50}) [S1].

Figure X. General ranges of response to radiation exposure by herbaceous plants as a percentage of the lethal dose (LD_{100}) [S3]



169. The Committee, in the UNSCEAR 1996 Report [U4], noted that protraction of radiation exposures increased the total doses required to kill plants [S4].

170. A range of sensitivities to radiation exposure was exhibited by the components of plants, ranging from dry seed (least sensitive) to apical meristems (most sensitive). Various crop plants showed different reductions in yield following radiation exposures, with further modifications being caused by external factors (e.g. temperature and humidity).

171. Plant species also varied in their tolerance to chronic radiation exposures. For the more sensitive pine species, dose rates of more than 3 mGy/h over 3–4 years reduced needle growth; in one-year-old saplings, needle length was substantially reduced when subjected to a dose rate of 7 mGy/h over a single growing season. Trunk growth was reduced in mature pine trees by dose rates in the range 0.4–2 mGy/h over a 9-year period. Delayed bud burst and an extended period of leafing out was demonstrated in white oaks chronically exposed to gamma radiation. At dose rates greater than 4 mGy/h, the trees were more susceptible to aphid infestation.

172. In view of the effects on the most sensitive plants evident with chronic exposure at dose rates of 1–3 mGy/h and of some specific changes noted at dose rates of 0.4–2 mGy/h, the Committee [U4] suggested that chronic dose rates at or below 400 μGy/h (10 mGy/d) should have only slight effects on sensitive plants but would be unlikely to produce any significant deleterious effects on the wider range of plants present in natural plant communities.

2. Terrestrial animals

173. The effects of radiation exposure on mammals had been extensively studied in radiobiological experiments using laboratory animals (mice, rats, dogs and monkeys) and domestic livestock (pigs, sheep, goats, burros and cattle) [B7, B8]. Except in the case of exposure involving unusually high doses, radiation damage or lethality in mammals results from disturbances in the haematopoietic system and the gastrointestinal mucosa. These cell self-renewal systems contain stem cells, differentiating cells and functional end cells, with the stem cells being the most radiosensitive and thus having the predominant influence on the radiation response. Symptoms become apparent when end cells are not replaced.

174. Protraction of a given total exposure generally reduces the extent of injury, as it allowed two distinct processes to intervene. First, sublethal damage is reparable at the cellular level, which is particularly important for exposures to low-LET radiation. Secondly, cell proliferation could replace lethally damaged cells and maintain the cell population at a new level, which is determined by the dynamic interaction between the dose rate and the rate of cell death, and by the total reserve proliferative capacity.

175. The Committee noted that at reduced dose rates (protraction of a given total dose) of low-LET radiation, all species showed a gradual increase in LD_{50}, i.e. higher total doses were tolerated. This changing response was attributed to the increasingly effective influence of cellular repair of sublethal damage at the lower dose rates. As the dose rate was further reduced, a sharply increasing trend in the values for the median lethal dose was apparent for mice, pigs, dogs, goats and sheep; the approximate threshold dose rates for this change in response corresponded to the accumulation of an LD_{50} dose within periods ranging from 0.2 days (mouse) to 9 days (goat). This rapid change in LD_{50} with dose rate was interpreted as being the consequence of a shifting balance in the dynamic interaction between the dose-rate-dependent cell loss and the cell proliferation and maturation kinetics in the haematopoietic system; the latter processes are under homeostatic control, i.e. their rate constants can alter in response to radiation-induced cell loss. The data for the burro (donkey) and primates did not show any sharp increase in the median lethal dose at dose rates down to 8.3 mGy/h (LD_{50} in 18 days) and 5.4 mGy/h (LD_{50} in 60 days), respectively. There did not appear to be any data for LD_{50} values at dose rates of less than 4 mGy/h or for exposure periods exceeding 60 days, although studies had been made outside these levels for other purposes.

176. While acknowledging that the numbers of mammalian species that had been (or indeed were likely to be) studied were extremely limited and probably atypical, the Committee noted [U4] that, even taking account of substantial interspecific variability, the available data provided very little evidence that chronic dose rates below 400 μGy/h (approximately 10 mGy/d) to the most exposed members of the population would seriously affect their mortality (and, thus, the death rate in populations of these species) from either deterministic or stochastic responses.

177. The effects of radiation exposure on reproduction had also been much studied, with most of the results suggesting that natality is a more radiosensitive parameter than mortality in species other than man and therefore of more relevance in an environmental context. The Committee considered that the minimum dose required to depress reproduction rates might be less than 10% of the dose required to produce direct mortality [W6].

178. The Committee suggested that damage to the developing mammalian embryo appeared to be a potentially significant criterion for assessing the impact of contaminant radionuclides in the natural environment. Dose rates of 420 μGy/h throughout gestation produced readily detectable reductions in the populations of germ cells in the developing gonads of a number of mammalian species, and the lowest dose rate at which damage had been seen was 10 μGy/h from tritium (as HTO in drinking water) incorporated in female mouse embryos. In addition, dose rates of the order of 420 μGy/h induced reductions in neonatal brain weight, although the significance of this deficit was unknown in functional or behavioural terms. The wider significance of these responses at the population level had not been investigated. Even recognizing that only very limited data were available, the Committee concluded that maximum dose rates of 100 μGy/h (2.4 mGy/d) to pregnant members of a mammalian population were unlikely to have any consequences for the population as a whole from the induction of damage in the developing embryos.

179. The Committee noted that the data on the radiosensitivity of terrestrial animals were dominated by data on mammals, the most sensitive class of organisms. Acute lethal doses ($LD_{50/30}$) were 6–10 Gy for small mammals and 1.5–2.5 Gy for larger animals and domestic livestock. When a total dose of magnitude similar to the $LD_{50/30}$ was delivered over a lifetime—for example, 7 Gy to the mouse (420 μGy/h, or 10 mGy/d)—the average loss of lifespan had been estimated to be about 5% and resulted from the induction of neoplastic disease [U9]. There was substantial inter-species variability, but, in general, little indication that dose rates below about 400 μGy/h to the most exposed individual would seriously affect mortality in the population.

180. The Committee noted that reproductive capacity was more sensitive to the effects of radiation exposure than life expectancy (mortality) and felt that the reproductive rates of mammals might be depressed at doses that were 10% of

those leading to mortality. It also felt that some loss of oocytes might occur at 1% of the lethal dose, but because of excess oocyte production, fecundity should be affected to a lesser extent. Mice, exposed from conception to a dose rate of 800 μGy/h, could be made sterile at 25 weeks. In the most sensitive mammal studied, the beagle dog, a dose rate of 180 μGy/h caused progressive cell depletion and sterility within a few months, but a dose rate of 36 μGy/h over the whole life produced no damaging response. The Committee concluded that a radiation dose rate of less than 40 μGy/h to the most exposed individual in a population (and most probably, therefore a lower mean dose rate to individuals in the population as a whole) would be unlikely to have an impact on the overall reproductive capacity of a mammalian population as a consequence of the effects of radiation exposure on fertility, fecundity or the production of viable offspring.

181. The effects of radiation exposure on birds had been shown to be similar to those on small mammals. Reptiles and invertebrates were less radiosensitive, although physiological differences began to make direct comparisons with other species less appropriate. The chronic exposure of one short-lived species of lizard in enclosures had shown no evident effects when exposed over 5 years at a dose rate of 830 μGy/h. In two longer-lived species of lizard, some individuals had been made sterile after 3.5 years at a dose rate of 630 μGy/h in one species and after 5.5 years at a dose rate of 210 μGy/h in another species. Adult invertebrates were seemingly quite insensitive to the effects of radiation exposure in terms of induced mortality, but the process of gametogenesis, developing eggs and juvenile stages were more sensitive.

3. Aquatic organisms

182. A number of reviews of the studies of the effects of exposure to ionizing radiation on aquatic organisms were available to the Committee [A3, B9, C3, E2, I2, I3, N1, N2, P2, T5, W9] during the preparation of the annex of the UNSCEAR 1996 Report [U4]. Some of these had been prepared specifically to provide a basis for assessing the potential effects of discharges of liquid radioactive effluents on aquatic organisms in their natural environment [I2, I3, N1, N2, W1].

183. Among aquatic organisms, fish were the most sensitive to the effects of radiation exposure; the developing fish embryos were particularly sensitive. The LD_{50} for acute irradiation of marine fish was in the range 10–25 Gy for assessment periods of up to 60 days following exposure. The upper end of the range of LD_{50} for marine invertebrates had been found to be several hundred grays. Embryos, on the other hand, were affected at much lower doses, for example, the $LD_{50/90}$ for salmon embryos was 0.16 Gy [B10].

184. Chronic exposures at dose rates of 10–30 mGy/h had no effect on the mortality of snails, marine scallops, clams and blue crabs. Dose rates somewhat above this range had some effects on food-limited populations of *Daphnia pulex*. Short-term (40 days) exposure of mosquito fish at dose rates in the range 14–54 mGy/h showed no radiation-induced mortality, but, for the closely related guppy, there was some indication that long-term exposure (>470 days) at dose rates above 1.7 mGy/h reduced the normal lifespan, particularly for males.

185. Reproductive effects are a more sensitive indicator of radiation response for aquatic organisms. Chronic dose rates in the range 3.2–17 mGy/h reduced the reproductive capacity in the freshwater snail, *Physa heterostropha*, and in the marine polychaete worms, *Ophriotrocha diadema* and *Neanthes arenaceodentata*. Exposure at a dose rate of 7.3 mGy/h rendered male freshwater fish (*Ameca splendens*) effectively sterile after 50 days, and exposure at a dose rate of 1.7 mGy/h over the lifespan of pairs of guppies (the freshwater fish, *Poecilia reticulata*) significantly reduced the lifetime production of offspring [W7]. It had been concluded that significant effects on fish gonads from chronic radiation exposure would be unlikely at dose rates less than 1 mGy/h [I3, W1]. Overall consideration of the data available led to the conclusion that chronic irradiation at dose rates up to 400 μGy/h to a small proportion of the individuals in an aquatic population (and, therefore, with correspondingly lower average dose rates to the whole population) would not have any detrimental effects at the population level [I4, N1].

C. Effects of radiation exposure on populations of plants and animals

186. The Committee noted in the annex of the UNSCEAR 1996 Report [U4] that one of the difficulties in evaluating the effects of radiation exposure on populations and ecosystems was the determination of the parameters to measure. Typically measured attributes at the population level included numbers of individuals, mortality rate, reproduction rate and mean growth rate. The Committee also noted that measurable changes in populations and communities required rather severe effects to be induced at the cellular and individual organism levels [e.g. W8]. The Committee also noted that genetic or somatic mutations that could be produced by relatively low levels of exposure might have little or no impact on population or community performance because of natural selection [B10, C4, M2, P3, T5] and the convergence of genetic information among adjacent populations [R1, T5].

187. The Committee also noted that the effects of radiation exposure at the population and community levels were manifest as a combination of direct changes due to radiation damage and indirect responses to the direct changes. This seriously complicated the interpretation of the effects of radiation exposure on organisms in the natural environment. The wide range of radiosensitivities of the organisms that make up most natural communities creates a situation where, if doses are such that the sensitive species, but not the more resistant ones, are affected, the latter might gain a significant competitive advantage and increase in abundance or vigour.

This could erroneously be interpreted as a hormetic response; such a response might not however be produced if the resistant species alone were irradiated. This is but one of many examples of indirect response to the direct effects of radiation exposure.

188. Because of the compensation and adjustment possible in animal species, the Committee considered that it is unlikely that radiation exposures causing only minor effects on the most exposed individual would have significant effects on the population. Reproductive changes are a more sensitive indicator of the effects of radiation exposure than mortality, and mammals are the most sensitive animal organisms. On this basis, chronic dose rates of less than 100 μGy/h to the most highly exposed individuals would be unlikely to have significant effects on most terrestrial animal communities. The Committee also concluded that maximum dose rates of 400 μGy/h to a small proportion of the individuals in aquatic populations of organisms would not have any detrimental effect at the population level. These conclusions referred to the effects of low-LET radiation exposure. Where a significant part of the incremental radiation exposure comes from high-LET radiation (alpha particles), the Committee considered that it is necessary to take account of the different RBEs.

D. Effects of major accidents

189. The UNSCEAR 1996 Report [U4] discusses the effects of two accidents in the former Soviet Union (at Chernobyl and at Mayak in the south-eastern Urals) leading to major releases of radioactive material into the environment [A28, G19, I23, I24, K1, K22, K23, N9, S29, S34, S40, T4, T27]. These accidents provided opportunities to observe radiation-related changes in plant and animal communities. The Committee noted however that any major accident is likely to be unique in terms of the quantity and composition of the radioactive material released, the time course of the release, the dispersal and deposition patterns, which are governed by local and regional meteorological or hydrological conditions, and the biochemical and geochemical character of the areas subject to contamination. Where long-lived radionuclides are released, biochemical and geochemical processes would determine the long-term behaviour and redistribution of the radionuclides in the environment. Given this multiplicity of factors, any major nuclear accident would be expected to yield new radioecological information. However, the primary concern following an accidental release of radionuclides is to ensure that the radiation risks to human populations are controlled and minimized. Consequently, the only environmental information likely to be collected is that which is immediately necessary to meet this objective. Such information is unlikely to be sufficient for the purposes of developing a complete radioecological description of the situation. The larger the incident and the greater its potential human impact, the more limited would be the resources available to collect radioecological information, particularly in the early phase following the accident.

190. In particular, the data required to develop estimates of the radiation exposure of wild organisms (i.e. the space and time-dependent variations of the radionuclide concentrations, especially of the short-lived radionuclides both within the organisms and in their external environment immediately following an accident) would not be known. These variations would result in substantial intra-species and inter-species inhomogeneities in exposure and would pose considerable difficulties for establishing a clear and reliable relationship between cause (the accumulated radiation dose) and any observed effect. In practice, it is likely that estimates of the dose rates in the early period following the release would be calculated subsequently from the observed distribution of deposition densities of the longer-lived radionuclides, from a knowledge of the relative quantities of the radionuclides released, and using models of radionuclide behaviour in the environment. Such dose-rate estimates are inevitably imprecise and could be subject to significant systematic error.

191. The highly variable habits and target geometries of the wild organisms are additional complicating factors. These range, for example, from soil bacteria to single-celled algae and protozoa, and include a wide variety of terrestrial and aquatic invertebrates, mammals (ranging from shrews to deer) and large deciduous or evergreen trees. Plants provide a very high surface area to mass ratio (compared with animals) for deposition/adsorption of a radioactive aerosol. Because the leaves, flowers and terminal buds of plants are responsible for energy absorption, growth and reproduction, a coincidence arises between radionuclide accumulation (and hence radiation dose) and potential radiosensitivity. Other examples of coincidence are the surface litter layer and its populations of invertebrate decomposers in terrestrial environments, and surface sediments and benthic organisms in aquatic systems.

192. Depending on the quantities of specific radionuclides released following an accident, the radiation exposures might range from low (a few multiples of the natural background) to high (absorbed doses greater than 1 Gy). Different phases of biological response to the higher total doses might be distinguished. Initially, and, in particular if short-lived radionuclides made up a significant proportion of the release, there might be an acute phase in which total doses sufficient to produce immediate or relatively early detectable biological responses are accumulated. In the intermediate phase, dose rates would decrease owing to the decay of the short-lived radionuclides and possibly, but not necessarily, owing to the redistribution of the longer-lived radionuclides by natural processes. Even in this phase, the slower accumulation of radiation dose might still result in total integrated doses sufficient to prevent recovery of organisms damaged in the initial phase or lead to the appearance of medium-term damage. In the long-term phase, post-irradiation recovery (and adaptation) becomes apparent, provided that the initial and medium-term damage had not been large enough to radically alter the population or community structure.

III. SUMMARY OF DOSE–EFFECTS DATA FROM THE CHERNOBYL ACCIDENT

193. A great deal of scientific information concerning the effects of exposure to ionizing radiation has been developed from studies of non-human biota in the area surrounding the site of the Chernobyl accident. The follow-up studies provided the main source of new information on the effects of radiation exposure on non-human biota since the UNSCEAR 1996 Report [U4]. This area has a temperate climate and flourishing flora and fauna. Much of the new information, originally reported in Russian, has been summarized in a report prepared for the Committee [A5] and by the work of the Chernobyl Forum [E8]. The following discussion of radiation levels and effects on biota observed in the region around the Chernobyl nuclear power plant is based on information presented in reference [E8] and in other recent reviews [G26].

A. Radiation exposure

194. The Chernobyl Forum Expert Group on Environment (EGE) [E8] noted that the effects of the Chernobyl accident should be studied within specific time periods. Three distinct phases of radiation exposure have been identified in the area local to the accident [U4]. In the first 20 days, radiation exposures were essentially acute because of the large quantities of short-lived radionuclides present in the passing cloud (^{99}Mo, ^{132}Te/I, ^{133}Xe, ^{131}I and ^{140}Ba/La). Most of these short-lived, highly radioactive nuclides deposited onto plant and ground surfaces, resulting in gamma radiation dose rates of up to about 20 Gy/d. However, for surface tissues and small biological targets (e.g. mature needles and the growing buds of pine trees) there was a considerable additional dose rate due to the beta radiation from the deposited radionuclides. High doses to the thyroids of vertebrate animals also occurred during the first days/weeks following the accident owing to the inhalation and ingestion of radioactive isotopes of iodine and their radioactive precursors.

195. The second phase of radiation exposure extended through the summer and autumn of 1986, during which time the short-lived radionuclides decayed and the longer-lived radionuclides were transported to different components of the environment by physical, chemical and biological processes. Dominant transportation processes included rain-induced transfer of radionuclides from plant surfaces onto soil, and bioaccumulation through plant tissues. Dose rates at the soil surface declined to much less than 10% of the initial values owing to radioactive decay of the short-lived radionuclides, but damaging total doses were still accumulated. Approximately 80% of the total radiation dose accumulated by plants and animals was received within 3 months of the accident, and over 95% of this was due to beta radiation exposure [E8]. Measurements made with thermoluminescent dosimeters on the soil surface at sites within the 30-km exclusion zone indicated that the ratio of beta to gamma dose was about 26:1, (i.e. 96% of the total dose was due to beta radiation exposure) [P18].

196. The EGE [E8] also defined a third (and continuing) phase of radiation exposure with chronic dose rates less than 1% of the initial values and derived mainly from ^{137}Cs. With time, the decay of the short-lived radionuclides and the migration of much of the remaining ^{137}Cs into the soil meant that the contributions to the total radiation exposure from the beta and gamma radiations tended to become more comparable. Reference [E8] noted that the balance depended on the degree of bioaccumulation of ^{137}Cs in organisms and the behaviour of the organism in relation to the main source of external exposure resulting from the ^{137}Cs in the soil.

B. Effects of radiation exposure on plants

197. The report of the EGE was a great advance on previous publications describing the follow-up work on the effects of the Chernobyl accident. In particular, the report gave considerable attention to evaluating the dosimetry of, and consolidating the information on the effects on non-human biota. Thus, given both the greatly improved quality of the data and the comprehensive nature of the evaluation provided by the EGE, much of the following discussion is adapted from reference [E8].

198. Doses received by plants arising from the deposited radionuclides resulting from the Chernobyl accident were influenced by the physical properties of the various radionuclides (i.e. their half-lives, radiation emissions, etc.), the physiological stage of the plant species at the time of the accident, and the different species-dependent propensities to take up radionuclides into critical plant tissues [E8]. The occurrence of the accident in late April 1986 was thought to have enhanced the damaging effects of the deposition because it coincided with the period of accelerated growth and reproduction of plants.

199. The deposition of beta-emitting radionuclides onto critical plant tissues resulted in their having received a significantly larger dose than animals living in the same environment [P18, P19]. According to reference [G9], large apparent inconsistencies in the dose–response observations occurred when the beta-irradiation component was not appropriately taken into account.

200. Within the 30-km zone around the Chernobyl plant, the doses to plants associated with the deposition of total beta activity (0.7–3.9 GBq/m^2) were sufficient to cause short-term sterility and reduction in productivity of some species [P19]. By August 1986, crops that had been sown prior to the accident began to emerge. Growth and development problems were observed in plants in fields with deposition densities of 0.1–2.6 GBq/m^2 of total beta activity, and with estimated dose rates initially received by the plants having reached 300 mGy/d. Spot necroses on leaves, withered tips of leaves, inhibition of photosynthesis, transpiration and metabolite

synthesis were detected, as well as an increased incidence of chromosome aberrations in meristem cells [S22]. The frequency of various anomalies in winter wheat exceeded 40% in 1986–1987, with some abnormalities apparent for several years afterwards [G12].

201. Coniferous trees were already known to be among the more radiosensitive plants, and the pine forests, 1.5–2 km west of the Chernobyl nuclear power plant, received sufficient doses, more than 80 Gy, at dose rates that exceeded 20 Gy/d, to cause mortality [T18]. The first signs of radiation injury were yellowing and needle death in pine trees in close proximity to the nuclear power plant and appeared during the summer of 1986. The colour of the dead pine stands resulted in the forest being referred to as the "red forest".

202. Tikhomirov and Shcheglov [T18] and Arkhipov et al. [A11] found that mortality rate, reproduction anomalies, stand viability, and re-establishment of pine-tree canopies were dependent on absorbed dose. Acute irradiation of *Pinus silvestris* at doses of 0.5 Gy caused detectable cytogenetic damage; at doses of more than 1 Gy, growth rates were reduced and morphological damage occurred; and, at more than 2 Gy, the reproductive abilities of trees were altered. Doses of less than 0.1 Gy did not cause any visible damage to the trees. Table 23 shows the variation in activity concentration and dose among pine trees within the 30-km zone. The radiosensitivity of spruce trees was observed to be greater than that of pines. At absorbed doses as low as 0.7–1 Gy, spruce trees had malformed needles, buds and shoot growth [K1].

203. About 90% of the absorbed dose to critical parts of the trees was due to beta irradiation from the deposited radionuclides with the remaining 10% from gamma irradiation. Table 22 summarizes the external gamma dose rates and the internal radionuclide concentrations in the conifers around the Chernobyl plant. By 1987, recovery processes were evident in the surviving tree canopies and the forests were re-establishing themselves where the trees had perished [A11]. In the decimated pine stands, a sudden invasion of pests occurred that later spread to adjoining areas. Grassland, with a slow invasion of self-seeding deciduous trees, has now replaced the deceased pine stands. Four distinct zones of radiation-induced damage to conifers were discernable (table 23).

Table 22. Activity concentration in needles of coniferous trees and estimated external gamma dose rates in October 1987 as a function of distance from the Chernobyl nuclear power plant

For azimuth 205 to 260 degrees (adapted from reference [K12])

Distance from NPP (km)	*External exposure rate (μGy/h)*[a]	*Accumulated external dose (mGy)*[a]	*Activity concentration in needles (kBq/kg)*					
			^{144}Ce	^{106}Ru	^{95}Zr	^{95}Nb	^{134}Cs	^{137}Cs
2	2 500	126 000	13 400	4 100	800	1 500	1 500	4 100
4	120	5 000	150	60	8	15	17	72
16	0.4	14	1.5	0.6	0.1	0.17	0.18	0.55

[a] Based on gamma radiation levels at 1 m height above the soil surface. The values given in the original reference were in mR/h and have been converted assuming 1 mR/h is equivalent to 10 μGy/h.

Table 23. Zones and corresponding damage to coniferous forest in the area around the Chernobyl nuclear power plant (from reference [K1])

Zone and classification	*External gamma dose (Gy)*	*Exposure rate (μGy/h)*[a]	*Internal dose to needles (Gy)*
Conifer death (4 km²) Complete death of pines Partial damage to deciduous trees	over 80–100	over 5 000	over 100
Sublethal (38 km²) Death of most growth points Partial death of coniferous trees Morphological changes to deciduous trees	10–20	2 000–5 000	50–100
Medium damage (120 km²) Suppressed reproductive ability Dried needles, morphological changes	4–5	500–2 000	20–50
Minor damage Disturbances in growth, reproduction and morphology of coniferous trees	0.5–1.2	<200	<10

[a] The values given in the original reference were in mR/h and have been converted assuming 1 mR/h is equivalent to 10 μGy/h.

C. Effects of radiation exposure on soil invertebrates

204. Between 60% and 90% of the initial fallout of radionuclides was captured by the forest canopy and other plants [E8]; however, within weeks to a few months, the processes of wash-off by rain and leaf fall removed most of the initial deposition to the litter and soil layers, where soil and litter invertebrates were exposed to high radiation levels for protracted time periods. The timing of the accident coincided with the most radiosensitive life stages of the soil invertebrates: reproduction and moulting following their winter dormancy [T18]. Within two months after the accident, the numbers of invertebrates in the litter layer of forests 3–7 km from the nuclear power plant were reduced by a factor of 30 [K11], and reproduction was strongly impacted (larvae and nymphs were absent). These effects corresponded to doses of approximately 30 Gy (estimated from TLDs placed in the soil) resulting in mortality of eggs and early-life stages, as well as reproductive failure in adults. However, within a year of the accident, reproduction of invertebrates in the forest litter resumed, due, in part, to the migration of invertebrates from less contaminated sites. After 2–3 years, the ratio of young to adult invertebrates in the litter layer, as well as the total mass of invertebrates per unit area, were no different from those in control sites; however, species diversity remained markedly lower [K11]. As noted in the report of the EGE [E8], this is important since the diversity of invertebrate species within the soil facilitates an analysis of the community-level effects of radiation exposure (i.e. changes in species composition and abundance). For example, only five species of invertebrates were found in 10 soil cores taken from pine stands in July 1986, 3 km from the Chernobyl nuclear power plant, compared to 23 species at a control site 70 km away. The mean density of litter fauna was reduced from 104 individuals per 225 cm^2 core at the control location to 2.2 at the 3-km site. Six species were found in all 10 cores taken from the control site, whereas no one species was found in all 10 cores from the 3-km location [K13]. The number of invertebrate species found in the heavily contaminated sites was only half that of controls in 1993, and complete species diversity did not recover until 1995, almost 10 years after the accident [K11].

205. A fourfold reduction in earthworm numbers was found in arable soils, but no catastrophic mortality in any group of soil invertebrates was observed. The dose to invertebrates in forest litter was 3–10 fold higher than that to those residing in unploughed surface soil since the radionuclides deposited on the surface had not migrated downwards. The result was no reduction in the numbers of soil invertebrates below a depth of 5 cm in the soil as they were shielded by the overlying soil [K11].

206. Although, the researchers were unclear if sterility of invertebrates occurred in the heavily contaminated sites around the Chernobyl nuclear power plant [K11], the 30 Gy cumulative dose reported in the field studies was within the range of experimental doses used to control pest insects by external irradiation. A recent review indicated that most insect, mite and tick families require a sterilization dose of less than 200 Gy [B40], although the sterilization dose for some insects and related arthropods is much lower than this and varies widely. As was found for plants [S2], the radiosensitivity of insects is related to the average interphase nuclear volume [B40].

D. Effects of radiation exposure on farm animals

207. Ruminants, both domestic (cattle, goats and sheep) and wild (elk and deer), generally receive relatively high doses in radioactively contaminated environments, because they consume large amounts of vegetation, and many radionuclides accumulate in their bodies. For example, a single cow consumes about 75 kg of fresh grass each day.

208. In the period shortly after the accident, domestic livestock within the 30-km zone were exposed to high levels of radioactive iodine (^{131}I and ^{133}I with half-lives of 8 days and 21 hours, respectively). This resulted in significant internal and external doses due to beta and gamma radiation exposure (table 24). A dose of about 76 Gy is sufficient to cause harm to the thyroid gland [B23]. Soils of Ukraine and Belarus are naturally low in stable iodine, cobalt and manganese. In conditions of endemic deficiency of stable iodine, the transfer of radioactive iodine from blood to the thyroid gland may be 2–3 times greater than normal [P19]. These conditions accentuated the consequences of the accident.

Table 24. Doses to cattle that stayed in the 30-km zone around the Chernobyl plant from 26 April to 3 May 1986 [K12]

Distance from nuclear power plant (km)	*Surface activity (10^8 Bq/m²)*	*Absorbed dose (Gy)*		
		Thyroid	*GI tract*	*Whole body internal*
3	8.4	300	2.5	1.4
10	6.1	230	1.8	1.0
14	3.5	260	1.0	0.6
12	2.4	180	0.7	0.4
35	1.2	90	0.4	0.2

209. Depressed thyroid function in cattle was related to the dose received (69% and 82% reductions in function with thyroid doses of 50 Gy and 280 Gy, respectively). The concentration of thyroid hormones in the blood of animals was lower than the physiological norm during the whole lactation period. Radiation damage to the thyroid gland was confirmed by histological studies (i.e. hyperplasia of connective tissue and sometimes adipose tissue, vascular hyperaemia and necrosis of epithelium). Animals with practically no thyroid tissue were observed in Ukraine. Disruptions of the hormonal status in calves born to cows with irradiated thyroid glands were especially pronounced [A12]. Similar effects were observed in cattle evacuated from the Belarusian portion of the 30-km zone [I18].

210. Although most livestock were evacuated from the area after the accident, several hundred cattle were maintained in the more contaminated areas for a 2–4 month period. By autumn 1986, some of these animals had died; others showed impaired immune responses, lowered body temperatures and cardiovascular disorders. Hypothyroidism lasted until 1989, and may have been responsible for reproductive failures in animals that received thyroid doses of more than 180 Gy [I18]. Offspring of highly exposed cows had reduced weight, reduced daily weight gains, and signs of dwarfism. Reproduction returned to normal in the spring of 1989. Haematological parameters were normal for animals kept in areas with ^{137}Cs deposition densities of 0.2–1.4 MBq/m^2 (5–40 Ci/km^2) [A12].

211. No increase in the rates of birth defects were detected above background levels at annual doses below about 0.05 Gy [P17].

E. Effects of radiation exposure on other terrestrial animals

212. Surveys and autopsies of wildlife and of abandoned domestic animals that remained within 10 km of the Chernobyl nuclear power plant were conducted four months after the accident. [K11]. Fifty species of birds were identified, including some rare ones; all appeared normal in appearance and behaviour. No dead birds were found. Swallows and house sparrows were found to be producing progeny that also appeared normal. Forty-five species of mammals from six orders were observed and no unusual appearances or behaviours were noted.

213. In a review of thirty-three studies of the biological consequences of the Chernobyl accident, Møller and Mousseau [M19] commented on various increases in mutations and cytogenetic abnormalities attributed to elevated radiation levels. They noted that the fitness consequences of such increases were largely unknown and cited a study of differences in phenotypes in barn swallows from near Chernobyl and those from relatively uncontaminated control areas [M18]. The authors suggested that mutations with slightly negative fitness effects could have been exported from the contaminated zones and potentially affected unexposed populations. In an exchange of views, Møller et al. [M17, M20] challenged the hypothesis of Smith [S26] that the impacts on barn swallows arose from factors other than radiation exposure, namely the change in habitat and wildlife community arising from changes in agricultural practices resulting from efforts to reduce the spread of radioactively contaminated food. Smith however noted that the most contaminated sites were located within abandoned lands, which had large differences in both land use and ecology from the control sites.

214. Some wildlife and domestic animals were shot and autopsied in August and September 1986. Dogs and chickens showed signs of chronic radiation syndrome (reduced body mass; reduced fat reserves; increased mass of lymph nodes, liver and spleen; haematomas present in liver and spleen; and thickening of the lining of the lower intestine). No eggs were found in the nests of chickens, nor in their ovaries.

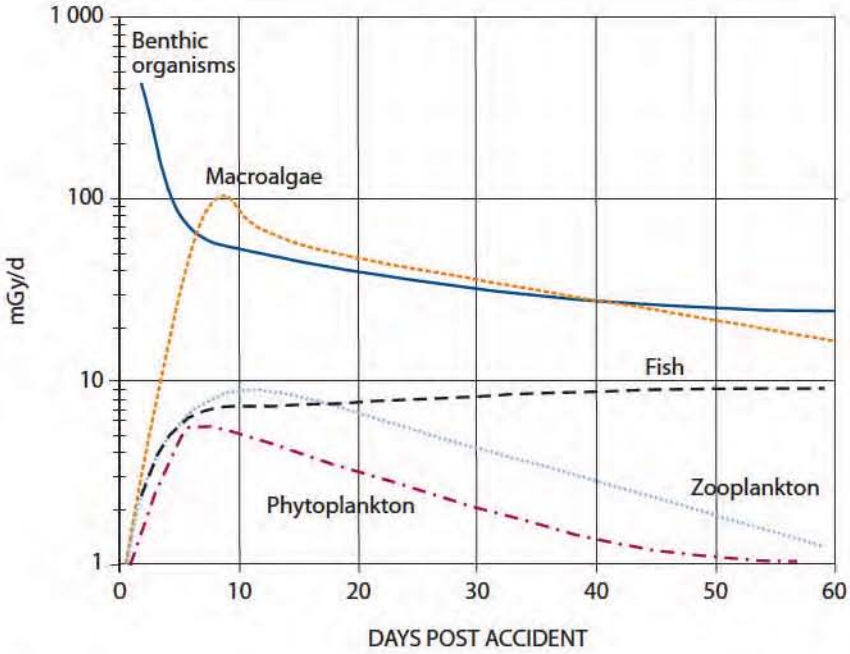
215. During the autumn of 1986, the number of small rodents on highly contaminated research plots decreased by a factor of 2–10. Estimates of absorbed doses during the first five months after the accident ranged from 12–110 Gy for gamma and 580–4,500 Gy for beta irradiation. By the spring of 1987, the numbers of animals were recovering, mainly due to immigration from less affected areas. In 1986 and 1987, the percentage of pre-implantation deaths in rodents in the highly contaminated areas was 2–3 fold greater than that in the controls. Resorption of embryos also increased markedly in rodents from the impacted areas; however, the number of progeny per female did not differ from that of the controls [T16].

F. Effects of radiation exposure on aquatic organisms

216. Cooling water for the Chernobyl nuclear power plant was obtained from a 21.7 km^2 man-made reservoir located to the south-east of the plant site. The cooling reservoir became heavily contaminated following the accident with a total activity of over 6.5 ± 2.7 PBq of a mixture of radionuclides (alpha and beta emitters) in the water and sediments [K14]. Aquatic organisms were exposed to external radiation from the radionuclides in the water, contaminated bottom sediments, and aquatic plants. Internal irradiation occurred as organisms took up radionuclides in their food and water or inadvertently consumed contaminated sediments. The resultant doses to aquatic biota over the first 60 days following the accident are depicted in figure XI.

Figure XI. The dynamics of absorbed dose rate to organisms within the Chernobyl nuclear power plant cooling pond during the first 60 days following the accident

Data are model results based on concentrations of radionuclides in the water column and lake sediments (adapted from reference [K12])



217. The maximum dose rates to aquatic organisms (excluding fish) were reported in the first two weeks after the accident, when short-lived radionuclides (primarily ^{131}I) contributed 60–80% of the dose. During the second week, the contribution of short-lived radionuclides to the doses of aquatic organisms decreased by a factor of two. Maximum dose rates to fish were delayed (see figure XI) owing to the time required for their food webs to become contaminated with longer-lived radionuclides (largely $^{134,137}Cs$, $^{144}Ce/Pr$, $^{106}Ru/Rh$ and $^{90}Sr/Y$). The dose rates to fish depended on their trophic positions. Non-predatory fish (carp, goldfish and bleak) incurred estimated peak dose rates of 3 mGy/d due to internal exposure in 1986, followed by significant reductions in 1987. Dose rates to predatory fish (perch), however, increased in 1987 and did not start to decline until 1988 [K12]. Accumulated doses were greatest for the first generation of fish born in 1986 and 1987. Bottom-dwelling fish (goldfish, silver bream, bream and carp) that were significantly irradiated by the bottom sediments accumulated total doses of approximately 10 Gy.

218. The reproductive capacity of young silver carp was analysed in 1990 [R10]. The fish were in live boxes within the cooling pond at the time of the accident. By 1988, the fish had reached sexual maturity. Over the entire post-accident period, they received a dose of 7–8 Gy. Biochemical analyses of muscles, liver and gonads indicated no difference from the controls. The amount of fertilized spawn was 94%; 11% of the developing spawn was abnormal. Female fertility was 40% higher than that of the controls, but 8% of the irradiated sires were sterile. The level of fluctuating asymmetry in offspring did not differ from that of the controls, although the level of cytogenetic damage (22.7%) significantly exceeded that of controls (5–7%). In contrast, Pechkurenkov [P20] reported that the number of cells with chromosome aberrations in 1986–1987 in carp, bream flat and silver carp was within the norm. It is worth noting that the cooling pond was subjected not only to radioactive contamination, but also to chemical pollution. Table 25 provides a summary of the recent reviews of the chronic effects of ionizing radiation exposure on the reproduction in fish. The Chernobyl accident data are included.

Table 25. Chronic effects of exposure to ionizing radiation on reproduction in fish

Derived from the FASSET database [C11]

Dose rate (μGy/h)	*Dose rate (mGy/d)*	*Reproductive effects*
0–99	0–2.4	Background dose group, normal cell types, normal damage and normal mortality observed
100–199	2.4–4.8	No data available

Dose rate (μGy/h)	*Dose rate (mGy/d)*	*Reproductive effects*
200–499	4.8–12	Reduced spermatogonia and sperm in tissues
500–999	12–24	Delayed spawning, reduction in testis mass
1 000–1 999	24–48	Mean lifetime fecundity decreased, early onset of infertility
2 000–4 999	48–120	Reduced number of viable offspring Increased number of embryos with abnormalities Increased number of smolts in which sex was undifferentiated Increased brood size reported Increased mortality of embryos
5 000–9 999	120–240	Reduction in number of young fish surviving to 1 month of age Increased vertebral abnormalities
>10 000	>240	Inter-brood time tends to decrease with increasing dose rate Significant reduction in neonatal survival Sterility in adult fish Destruction of germ cells within 50 days in medaka fish High mortality of fry, germ cells not evident Significant decrease in number of male salmon returning to spawn; after 4 years, female salmon had significantly reduced fecundity

G. Genetic effects in animals and plants

219. High quality data on the incidence of radiogenic mutations in plants and animals as a result of the accident are relatively sparse. An increased mutation level was apparent in 1987 in the form of various morphological abnormalities in Canada fleabane, common yarrow and mouse millet. Examples of abnormalities included: unusual branching of stems; doubling the number of racemes; abnormal colour and size of leaves and flowers; and development of "witch's broom" in pine trees. Similar effects within 5 km of the nuclear power plant also appeared in deciduous trees (leaf gigantism, and changes in leaf shapes). Morphological changes were observed at an initial gamma dose rate of 4.2–6.3 mGy/d. At a dose rate of 15.8–31.5 mGy/d, enhancement of vegetative reproduction (in heather) and gigantism of some plant species were observed [A11, K10, T17, T18].

220. Cytogenetic analysis of cells from the root meristem of winter rye and wheat germ of the 1986 harvest demonstrated a dose dependency in the number of aberrant cells. A significant excess over the control level of aberrations was observed at an absorbed dose of 3.1 Gy. Inhibition of mitotic activity occurred at a dose of 1.3 Gy, and germination was reduced at a dose of 12 Gy [G10]. The analysis of three successive generations of winter rye and wheat on the most contaminated plots revealed that the rates of aberrant cells in the intercalary meristem in the second and third generations were higher than in the first.

221. From 1986–1992, mutation dynamics were studied in populations of *Arabidopsis thaliana Heynh.* (L.) within the 30-km zone [A10]. On all study plots during the first 2–3 years after the accident, *Arabidopsis* populations exhibited an increased mutation burden. In later years, the level of lethal mutations declined; nevertheless the mutation rate in 1992 was still 4–8 times higher than the spontaneous level. The dose dependence of the mutation rate was best approximated by a power function with an exponent value of less than one.

222. Zainullin et al. [Z2] observed elevated levels of sex-linked recessive lethal mutations in natural *Drosophila melanogaster* populations living under conditions of increased chronic exposure to radiation resulting from the Chernobyl accident. The mutation levels were increased during 1986–1987 in flies inhabiting the more contaminated areas with initial exposure rates of 2 mGy/h (expressed as 200 mR/h in the original text) and more. During the subsequent two years, mutation frequencies gradually returned to normal.

223. Shevchenko et al. [S21] and Pomerantseva et al. [P16] reported studies of adverse genetic effects in wild mice. These involved mice caught during 1986–1991 within a 30-km radius of the Chernobyl nuclear power plant with different levels of gamma radiation exposure and, during 1992–1993, on a site in the Bryansk Oblast, Russia. The estimated total doses of gamma and beta radiation varied widely; the dose rates reached 3–4 Gy per month in 1986–1987. One endpoint was dominant lethality, measured by embryo mortality in the offspring of wild male mice mated with unexposed female laboratory mice. The dominant lethality rate was elevated for a period of a few weeks following capture in mice sampled at the most contaminated site. At dose rates

of about 2 mGy/h, 2 of 122 captured males produced no offspring and were assumed to be sterile. The remainder showed a period of temporary infertility and reduced testis mass. Fertility and testis mass, however, recovered with time after capture.

224. The frequencies of reciprocal translocations in mouse spermatocytes were consistent with previous studies. A dose-rate-dependent incidence of increased reciprocal translocations (scored in spermatocytes at meiotic metaphase I) was observed in all collected mice. The frequency of mice harbouring recessive lethal mutations decreased with time after the accident [P16]. Radiation-related gene mutation is unlikely to have any adverse effect on populations, at the dose rates that prevail now.

225. Increasing sophistication in the technologies for the detection of molecular and chromosomal damage have allowed researchers on the genetic consequences of the Chernobyl accident to examine endpoints not previously considered [E8]. Most prominent, and controversial, is the technique involving the measurement of mutation frequencies in repeat DNA sequences termed "minisatellite loci" or "expanded simple tandem repeats" (ESTR). These are repeat DNA sequences that are distributed throughout the germline and have a high background (spontaneous) mutation rate. Presently, ESTRs are considered to have no function, although this is a matter of much interest and discussion [B33, C10, I9]. Minisatellite mutations have only rarely been associated with recognizable genetic disease.

226. Although laboratory examination of mutations in mouse ESTR loci show clear evidence of a mutational dose response [D4, F16], the EGE was not aware of any convincing data on elevated levels of minisatellite mutations in plants or animals residing in the contaminated areas having been published in peer-reviewed scientific literature [E8]. In general, quantitative interpretation of the ESTR data is difficult because of conflicting findings, their weak association with genetic disease, dosimetric uncertainties and methodological problems [C10]. This is an area of science that requires additional research.

H. Overall observations on the effects of the Chernobyl accident

227. According to the EGE [E8], prior to the accident, much of the area around the Chernobyl nuclear power plant was covered by 30–40 year old pine stands that, from a successional standpoint, represented mature, stable ecosystems. The high dose rates due to ionizing radiation exposure during the first few weeks following the accident altered the balance in the community and opened niches for immigration of new individuals.

228. The ecological conditions within the 30-km Chernobyl exclusion zone arose from the complex interaction of a number of factors. The highest level of contamination occurred within this zone. As a result of the elevated radiation doses associated with the contamination, human activities such as agriculture, forestry, hunting and fishing within the exclusion zone were stopped [E8]. After the accident, the fields continued to yield agricultural produce for a number of years and, in the absence of active management in the areas that had been evacuated, many animal species, especially rodents and wild boars, consumed the abandoned cereal crops, potatoes and grasses as an additional source of forage [E8]. This was advantageous to these animal species and, along with the special reserve regulations established in the exclusion zone (e.g. a ban on hunting), tended to compensate for the adverse biological effects of radiation exposure and promoted an increase in the populations of wild animals, including game mammals (wild boars, roe deer, red deer, elk, wolves, foxes, hares, beaver, etc.) and bird species (black grouse, ducks, etc.) [G8, S23]. In addition, the Chernobyl exclusion zone has become a breeding area of white-tailed eagles, spotted eagles, eagle owls, cranes and black storks [G9].

229. The high dose rates from ionizing radiation during the first few weeks following the Chernobyl accident affected the balanced community by killing sensitive individuals, altering reproduction rates, destroying some resources (e.g. pine stands), making other resources more available (e.g. soil water), and opening niches for immigration of new and sometimes negative organisms (e.g. negative entofauna). These components and more, were interwoven in a complex web of action and reaction that altered populations and communities of organisms [E8].

230. Overall, the EGE [E8, H25] arrived at a number of general observations from their evaluation of the Chernobyl data, namely that:

- Radiation from radionuclides released as a result of the Chernobyl accident caused numerous acute adverse effects on the biota located in the areas of highest exposure (i.e. up to a distance of a few tens of kilometres from the release point). Beyond the exclusion zone, no acute radiation-induced effects on biota have been reported;
- The environmental response to the increased radiation exposure incurred as a result of the Chernobyl accident was a complex interaction among radiation dose, dose rate and its temporal and spatial variations, as well as the radiosensitivities of the different taxons. Both individual and population effects caused by radiation-induced cell death were observed in plants and animals and included increased mortality of coniferous plants, soil invertebrates and mammals; reproductive losses in plants and animals; and chronic radiation sickness in animals (mammals, birds, etc.);
- No adverse radiation-induced effects were reported in plants and animals exposed to a cumulative dose of less than 0.3 Gy during the first month after the accident (i.e. <10 mGy/d, on average); and